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Executive summary

Biodiversity is an increasingly important element of ICES' work, and is one of the research topics of strategic importance identified in the ICES Science Plan. The European Commission's (EC) recent Marine Strategy Framework Directive (MSFD) highlights the importance of marine biodiversity, and so requests for information from ICES on the monitoring, assessment and integration of biodiversity information will undoubtedly increase in the future. A range of ICES Expert Groups are currently involved in various aspects of marine biodiversity, and WGBIODIV aims to provide the ICES community with an improved capacity to coordinate, integrate and synthesise biodiversity information.

Many earlier studies of the diversity of marine species have been derived from surveys using a single gear, whether this is collecting grab samples for benthic infauna or trawl samples for demersal fish. In recent years, however, there have been an increased number of multidisciplinary, ecosystem surveys to inform on the diversity of a broader spectrum of marine organisms. In Section 2 we summarise briefly some of the methods for examining the diversity across multiple groups, and review the spatial distribution of distinct faunal assemblages in parts of the ICES area. The latter topic was included, as biodiversity indicators in support of the Marine Strategy Framework Directive (MSFD) may be implemented for geographic subdivisions "*in order to take into account the specificities of a particular area*".

Given that multiple indicators may be required to support MSFD Descriptor 1 (related to biological diversity), WGBIODIV have given some consideration to indicator development (Section 3). Biodiversity is of too broad a scope to be monitored in its entirety, yet certain facets can be assessed. Issues discussed include the potential differences between monitoring biodiversity and biodiversity loss, the identification and prioritisation of species and habitats for biodiversity monitoring, relevant groups for monitoring, and the potential overlap between metrics to inform on 'biodiversity' as well as other descriptors within the MSFD. This section also comprises a case study of assessing the species diversity of North Sea fishes, including some of the caveats regarding survey data.

Several ICES nations have been mapping particular elements of biodiversity to inform on areas of high species richness (biodiversity hotspots), displaying patterns of biodiversity across regions or national waters, and to inform on spatial planning. Recent approaches to the spatial analysis of biodiversity information are in Section 4, with examples from Dutch and Belgian national waters.

Given the need for Member States to assess biodiversity, as well as other elements of the ecosystem that are identified in other MSFD descriptors, survey data from offshore areas will likely be an important source of relevant information. These surveys may not have been designed originally to inform on species diversity and so there are important limitations and caveats that need to be identified. Section 5 discusses many of the issues, including gear selection, site selection, density of sampling stations, sample replication, catch processing, taxonomic resolution, and data filtering and standardisation.

WGBIODIV, in conjunction with the ICES Strategic Initiative on Biodiversity (SIBAS), were asked to identify the potential capacity of the ICES science community to address important biodiversity science issues and to provide advice on these topics. Various biodiversity issues are discussed briefly (Section 6), and some of the areas

and disciplines for which the ICES community is well placed to comment are identified.

The recently reported Census of Marine Life has provided a platform for many international studies collecting information on many elements of marine biodiversity. Section 7 provides the reader with a brief overview of the various projects and initiatives that have been undertaken by this unique venture.

1 Introduction

1.1 Background

The Study Group on Biodiversity Science (SGBIODIV) first met in 2007 in Belgium (ICES, 2007), and reported on possible contributions by ICES on biodiversity science, especially in terms of how such knowledge on biodiversity science could be used in the Ecosystem Approach to Management (EAM). The following year SGBIODIV met again in Belgium (ICES, 2008) in order to define 'biodiversity science' and report on the remit of the group, to review current and emerging marine biodiversity initiatives, and to provide an overview of how other ICES Expert Groups contributed to biodiversity science.

In 2009, SGBIODIV met in Germany (ICES, 2009) in order to suggest possible options for the better integration of biodiversity science across the ICES science and advisory community. It was during this third meeting that the members of SGBIODIV considered that there was a strong rationale for the Study Group to be established as a Working Group, as this would "*enable biodiversity science to be delivered as an overarching theme in a more coordinated manner*" and so "*better enable ICES to answer questions on marine biodiversity and to synthesise biodiversity-related information as a basis for advice*".

In 2010, the group was re-named the Working Group on Biodiversity (WGBIODIV) and met in Lisbon, Portugal (ICES 2010). During this meeting, WGBIODIV provided an overview of the current field programmes that survey some of the major marine taxa across the ICES eco-regions, and highlighted some of the relevant advantages, limitations and caveats in terms of how such data can be applied to biodiversity science. It was highlighted that, although there is a long history of coordination across the ICES community for surveying the main marine fish species (e.g. through trawl surveys), the spatial and/or temporal extent for surveys examining other, non-target, marine taxa is often more limited. In terms of developing indicators of biodiversity, WGBIODIV also briefly reviewed some elements of macroecology that need to be better considered, as well as the variety of indices and metrics that may be considered for the development of 'biodiversity indicators' (e.g. species-specific metrics; traditional multi-species community/assemblage metrics; taxonomic diversity; functional diversity; size-based and food-web or trophic indicators).

For the purposes of this report, we retain the definition of biological diversity as that given under the Convention of Biological Diversity (CBD), which is "*the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems*".

As suggested in an earlier SGBIODIV report (ICES, 2008), biodiversity science and the remit of the group is defined as "*scientific research into the understanding, conservation, restoration and sustainable use of the marine biodiversity of the North Atlantic Ocean and adjacent seas*".

In terms of policy, two of the main driving forces for the assessment of biodiversity are the CBD and the Marine Strategy Framework Directive (MSFD).

In April 2002, the Parties to the CBD committed themselves to achieve by 2010 a "*significant reduction of the current rate of biodiversity loss at the global, regional and national level as a contribution to poverty alleviation and to the benefit of all life on Earth*".

The European Marine Strategy Framework Directive (MSFD), adopted in June 2008, emphasises that "*The marine environment is a precious heritage that must be protected, preserved and, where practicable, restored with the ultimate aim of maintaining biodiversity and providing diverse and dynamic oceans and seas which are clean, healthy and productive*" (CEC, 2008). The directive aims to achieve Good Environmental Status (GES) by 2020 and its major programme is biodiversity-related. Of the eleven defined qualitative descriptors for determining GES, one is specifically designated as an overarching indicator for biodiversity (MSFD descriptor 1) stating that "*Biological diversity is maintained. The quality and occurrence of habitats and the distribution and abundance of species are in line with prevailing physiographic, geographic and climatic conditions*", although several of the other descriptors are also clearly biodiversity-related (see Borja *et al.*, 2010 and Cochrane *et al.* 2010 for further discussion).

The EC has also recognised the importance that "*monitoring methods are consistent across the marine region or subregion so as to facilitate comparability of monitoring results*" (CEC, 2008). Consequently, ICES will likely be involved in ensuring standardised sampling (e.g. through the survey groups) and analyses of such data that may be undertaken by various ecology and other Expert groups within the ICES Community.

1.2 Terms of Reference

The **Working Group on Biodiversity** (WGBIODIV), chaired by Jim Ellis, UK, will be renamed **Working Group on Biodiversity Science** (WGBIODIV) and will meet at ICES HQ, Copenhagen, Denmark, 21–25 February 2011 to:

a) Further develop unified analyses of the diversity for multiple groups (e.g. invertebrate and fish) to better examine overall biodiversity, and to compare and contrast spatial-temporal patterns in 'biodiversity' across ecological groups, with reference to ecosystem function;

b) Further explore and assess potential biodiversity indicators, for example by undertaking comparative analyses of taxonomic, functional, surrogate and trophic metrics;

c) Review the existing spatial approaches in assessing biodiversity status, and the spatial and temporal scales on which different elements of marine biodiversity operate, with regards the implications for survey design and indicator development;

d) Examine the implications of survey design for estimating 'biodiversity metrics';

e) Liaise with the ICES Strategic Initiative on Biodiversity (SIBAS) to identify the potential capacity of the ICES science community to address key biodiversity science issues[1] and provide Biodiversity advice.

f) [1]The ICES capacity to address, inter alia the following issues should be discussed and reported:

- i) Biodiversity and ecosystem services: the role of biodiversity in supporting ecosystem services and the social and economic consequences of human impacts on biodiversity;
- ii) Diversity and ecological processes: The extent to which the diversity of a community influences (a) 'stability', (b) productivity, (c) resistance to invasion or disease, and (d) ability to recover from natural and human impacts, and interactions between these factors. The changes in production among systems that differ in biodiversity. The role of biological invasions in altering system production and energy flow.
- iii) State of biodiversity: patterns and trends in biodiversity and the structuring roles of evolution, ecology and environment.

iv) Functional significance of biodiversity: the functional significance of genetic, species, population and ecosystem diversity. Redundancy and the extent to which species in a functional group are interchangeable. Comparisons of system function and biodiversity.

v) Measuring biodiversity: measurements of genetic, species, and ecosystem biodiversity and the errors associated with these measurements. The effects of errors on understanding of ecosystem structure and function.

vi) Biodiversity futures: projecting future changes in biodiversity in response to projected human and environmental drivers

g) Consider the results of the recently completed Census of Marine Life (CoML) project in the context of the ICES Science Plan.

WGBIODIV will report by 31 March 2011 (via SSGEF) for the attention of SCICOM.

1.3 Participants

The following participants attended the meeting or contributed by correspondence (denoted *).

*Odd Bergstad	Norway
*Ángel Borja	Spain (Basque Country)
Oscar Bos	Netherlands
Anik Brind'Amour	France
Wenche Eikrem	Norway
Jim Ellis	UK (England & Wales)
Simon Greenstreet	UK (Scotland)
Åge Høines	Norway
Juan Pablo Pertierra	European Commission (Observer)
Maria Põllupüü	Estonia
Nikolaus Probst	Germany
Heye Rumor	Germany
Melanie Sapp	UK (England & Wales)
*Michaela Schratzberger	UK (England & Wales)
*Jan Vanaverbeke	Belgium
Francisco Velasco	Spain

1.4 Summary of Working Documents and presentations

Although no formal Working Documents were presented, there was a presentation on biodiversity hotspots in Dutch waters by O. Bos.

1.5 References

Borja, Á., M. Elliott, J. Carstensen, A.-S. Heiskanen, W. van de Bund, 2010. Marine management - Towards an integrated implementation of the European Marine Strategy Framework and the Water Framework Directives. Marine Pollution Bulletin, 60: 2175-2186.

Cochrane, S. K. J., D. W. Connor, P. Nilsson, I. Mitchell, J. Reker, J. Franco, V. Valavanis, S. Moncheva, J. Ekebom, K. Nygaard, R. Serrao Santos, I. Naberhaus, T. Packeiser, W. van de Bund, A. C. Cardoso, 2010. Marine Strategy Framework Directive – Task Group 1 Report Biological Diversity. EUR 24337 EN – Joint Research Centre, Luxembourg: Office for Official Publications of the European Communities: 110 pp.

European Commission. 2008. Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive).

ICES. 2007. Report of the Study Group on Biodiversity Science (SGBIODIV), 9–11 May. MHC:11; 31 pp.

ICES. 2008. Report of the Study Group on Biodiversity Science (SGBIODIV), 11–14 March 2008, Gent, Belgium. ICES CM 2008/MHC:06; 71 pp.

ICES. 2009. Report of the Study Group on Biodiversity Science (SGBIODIV), 17–20 March 2009, Wilhelmshaven, Germany. ICES CM 2009/MHC:05; 51 pp.

ICES. 2010. Report of the Working Group on Biodiversity (WGBIODIV), 22–26 February 2010, Lisbon, Portugal. ICES CM 2010/SSGEF:06; 97 pp.

2 Studies on the wider biodiversity of marine habitats

2.1 Introduction

The sea is spatially very diverse in terms of patterns of bathymetry, sediment, water stratification, currents and living organisms. It is also diverse over temporal scales as important natural processes can operate over periods ranging from minutes to decades. The complexity of natural processes results in a mosaic of environmental conditions. Knowing where and which areas display coincident or contrasting patterns of faunal and habitat diversity is essential to conservation and Marine Spatial Planning (MSP) (Nicholson *et al.* 2006).

The MSFD requires biodiversity-based indicators that will lead towards the achievement of Good Environmental Status (Cardoso *et al.* 2010; Cochrane *et al.* 2010; European Commission 2008, 2010). Identifying distributions and understanding the individual and combined diversity patterns of various groups of organisms is an important task in meeting the requirements of the MSFD. The identification of areas of coincident or contrasting patterns of biodiversity is thus a truly multivariate issue, where single groups do not fully indicate the overall diversity of a system.

Given the need to better inform on 'regional biodiversity' across multiple taxa, WGBIODIV proposed the following ToR "To further develop unified analyses of the diversity for multiple groups (e.g. invertebrate and fish) to better examine overall biodiversity, and to compare and contrast spatial-temporal patterns in 'biodiversity' across ecological groups, with reference to ecosystem function" (ICES 2010). In the present section, we have excluded the reference to the ecosystem function, as that subject is considered to some extent in Section 6, although this topic should be revisited by WGBIODIV in future meetings.

2.2 Defining coincident and/or contrasting patterns of diversity

Many traditional studies of marine species diversity have been based on taxa and/or gear specific metrics. For example, there are many studies on the structure, distribution and diversity of benthic (e.g. Warwick, 1984; Rees *et al.*, 2007), epibenthic (e.g. Jennings *et al.*, 1999; Zühlke *et al.*, 2001; Callaway *et al.*, 2002; Ellis *et al.*, 2002b) and fish assemblages (e.g. Bergstad *et al.*, 1999; Lekve *et al.*, 1999; Ellis *et al.*, 2002a; Jovanovic *et al.*, 2007; Menezes *et al.*, 2006; Neves *et al.*, 2008) of various parts of the ICES area.

However, there have been comparatively few studies that have examined the wider biodiversity of particular sites (e.g. across multiple taxa, using various sampling gears). For example, the Le Danois Bank in the Cantabrian Sea has been subject to multidisciplinary sampling (Sánchez *et al.*, 2005, 2008, 2009), Ellis *et al.* (2011) used Day grab, 2 m beam trawl and 4 m beam trawl to better understand the range of species (meiofauna, infauna, epifauna and demersal fish) that occur in sandbank habitats in the southern North Sea, and Pascual *et al.* (submitted) incorporated zooplankton, benthic flora, benthic fauna, fish, seabirds and cetaceans in evaluating the biodiversity of the Basque Country coast (Cantabrian Sea, Bay of Biscay).

The use of "integrated" ecosystem surveys and the need to broaden our view of the studied communities, to include target as well as non-target species at the bottom of the food web, will likely increase in the future and will be essential to meet legal obligations under, for instance, the MSFD (Brind'Amour *et al.* 2009). Such issues are addressed by the ICES Working Group on Integrating Surveys for the Ecosystem Approach (WGISUR).

Studies that include a sampling design broad enough to sample across higher taxa (fish, benthos etc.), with material identified at the species level, definitely require multidisciplinary skills and approaches. These approaches are cost effective in ship time, but time/cost consuming for sample processing, and so multi-taxa field studies are rarely undertaken over broad spatial/temporal scales. In some cases, however, several monitoring programs focusing on different groups are conducted in the same region. The data in such programs often come from several surveys conducted using different sampling designs (gear, temporal scales) that have been undertaken at different sampling locations (e.g. the French Channel Ground Fish Survey, the International Bottom Trawl Survey of the North Sea, and Eastern English Channel beam trawl survey overlap).

2.2.1 Methodological considerations on the use of diversity indicators

Studying, defining and comparing patterns of biodiversity underlies several assumptions. The aim here is to present and discuss methodological considerations for some of these assumptions.

As a first step towards the identification of coincident or contrasting patterns of diversity, it is worth to address benefits and drawbacks of "diversity indicators". That is, an indicator for which sufficient sampling effort was deployed in order to have an accurate estimate of that indicator. This is a major statistical concern for indicators that may be very informative but highly sample size-dependent, such as species richness.

Once an appropriate index is chosen one should keep mind that processes operating at various spatial and temporal scales underlie coincident or contrasting diversity patterns. Therefore, interpretations of group diversity patterns should be considered as tentative prior to comparisons. For instance, using fish data from the Water Framework Directive (WFD), Nicolas *et al.* (2010) compared the functional diversity of fish between 31 European tidal estuaries, from Portugal to Scotland. They highlighted several spatial patterns of species richness and functional diversity. These patterns were mostly related to system size and entrance width, salinity gradient, and proportions of certain habitat (e.g. intertidal mudflats).

Once indicators and trends or patterns are identified, the question of how to combine and integrate results arises. Various methods for aggregating diversity indices across higher taxa and habitats can be used: weighted or non-weighted sum, arithmetic or geometric mean of the index' values. In most cases, authors have used a simple sum (e.g. Williams *et al.* 1997) or average of the values without any weighting of the values. This is probably the simplest way to combine values with fewer preconceptions. However, not weighting the group values when aggregating them into an overall index of biodiversity also implies an assumption: that all taxa have the same ecological importance when evaluating the environmental status using diversity indicators.

The choice of multiplying the values instead of summing them has implication in terms of the precautionary principle. For instance, when scaled diversity values lie between 0 and 1, taxa with low values will have a great effect on the global scoring and can thus "penalize" the result of the environmental status assessment. Borja *et al.* (submitted) proposed to use biodiversity evaluation methods to integrate large amounts of information from several components of the ecosystem, from plankton to mammals, transforming the information in values from 0 to 1, as in the WFD.

The main drawbacks when aggregating diversity values are that i) the method used to aggregate the values is usually empirical and not founded on any ecological assumptions, and ii) data for individual groups of species are defined over different spatial and temporal scales. These methodological choices are typically the same as the ones made when combining several diversity indicators (e.g. species, functional, genetic diversity).

2.2.2 Ways to overcome some methodological drawbacks

Variation in species richness can also be assessed by combining higher-taxon richness for different taxa (e.g. family/subfamily-richness, Williams *et al.* 1997). Although families are subject to the same "combining" assumptions as are the species, they may be less sensitive to small sample size. Given that the family spatial distribution overcomes the species distribution, lower sampling error (i.e. random error) at the family level could be expected. Estimating diversity patterns at the family level might also be an indirect way of assessing (or as a surrogate for) wider genetic diversity (e.g. the contribution of monophyletic groups).

New and original methods to spatially combine indicators are obviously needed. Statistical approaches, originally developed in other fields of study (e.g. forestry management Dray *et al.* 2002), could be useful to overcome the problem of combining indicators derived from two or more datasets (e.g. fish, benthos, seabirds). Typically, approaches such as the three-table method where two datasets (e.g. fish diversity indices and benthic diversity indices) are joined through a spatial neighbourhood matrix could be very useful. Durieux *et al.* (2010) recently used such an approach in the Bay of Seine to find coincident patterns of fish and benthos distributions. Although they did not use diversity indices, the method could easily be adapted to determine co-occurrence of diversity patterns. Outputs of such methods are co-occurrence maps of diversity indices and identification of statistically significant correlations between the two datasets.

Pascual *et al.* (submitted) provided information for several ecosystem components and their integrative evaluation, together with the reliability of the results, taking into account the spatial and temporal availability of data (Derous *et al.*, 2007). Following an approach similar to that undertaken by Borja *et al.* (2009), when integrating ecological status at the water body level, Borja *et al.* (submitted) integrated the biodiversity evaluation into a unique value for the whole of the Basque continental shelf; this was a similar approach to the Ecological Quality Ratio (EQR) within the WFD. In this particular case, reference conditions for high values do not exist; and environmental targets, as required under the MSFD, can be used (see Borja *et al.* submitted). Such targets can guide progress towards achieving good environmental status.

2.3 Biodiversity patterns within the MSFD regions and subregions

The MSFD will operate over defined regions, namely the Baltic Sea, North-east Atlantic Ocean, Mediterranean Sea and Black Sea. Additionally, Member States may, "*in order to take into account the specificities of a particular area*", implement the MSFD by reference to subdivisions if these are "*delimited in a manner compatible with … marine subregions*". The subregions of the North-east Atlantic being (i) Greater North Sea, including the Kattegat, and the English Channel; (ii) Celtic Seas; (iii) Bay of Biscay and the Iberian Coast; (iv) in the Atlantic Ocean, the Macaronesian biogeographic region, being the waters surrounding the Azores, Madeira and the Canary Islands. WGBIODIV did not have the expertise to consider the latter area during the meeting.

It should also be recognised that in many ways the MSFD bounds with the WFD (Borja *et al.*, 2010), especially for those species and habitats that occur in marine and transitional waters. Both Directives aim to develop indicators to monitor and maintain good environmental status. A major difference between these two directives lies in the fact that the WFD implied intercalibration exercises in order to make the general environmental objective operational in a harmonised way throughout the EU. Indeed some of the challenges in the MSFD will be methodological-oriented.

Here a brief overview of some of the subregions and benthic and fish assemblages is provided, and this is designed to be a preliminary guide for identifying relevant sources of information that may be useful in ensuring that any subdivisions considered under the MSFD are biologically meaningful. For further information, the reader is referred to the report of the Working Group for Regional Ecosystem Description (WGRED, ICES 2008).

2.3.1 Baltic Sea

Ojaveer *et al.* (2010) recently assessed the biodiversity status of the Baltic Sea (Figure 2.1) although the underlying method was not fully explained (HELCOM 2009). Based on a variety of different source material (i.e., journal articles, published reports, grey literature, unpublished data), they estimated the total number of cyanobacteria, phytoplankton, zooplankton, phytobenthos, zoobenthos, fish, marine mammals, and bird species as well as vertebrate parasites inhabiting the Baltic Sea. Comparison of the different groups indicated coincident patterns, for instance, for five of the six groups analysed in the Kattegat. Detailed analysis of these groups suggested high benthic diversity in that part of the Baltic Sea, whereas the Gulf of Finland displayed greater pelagic (i.e. mid-water) diversity (Figure 2.2).

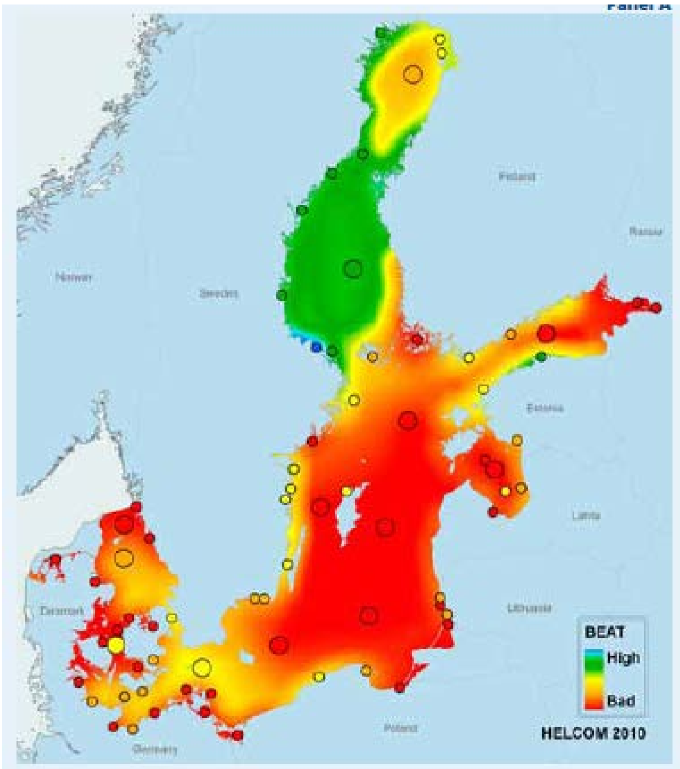

Figure 2.1. Preliminary integrated classification of biodiversity status of the Baltic Sea. Explanation on the methodology behind the status assessment is found in HELCOM (2009). There is a gradient ranging from unacceptable (warm colours) to acceptable biodiversity status. The circles represent the assessment sites.

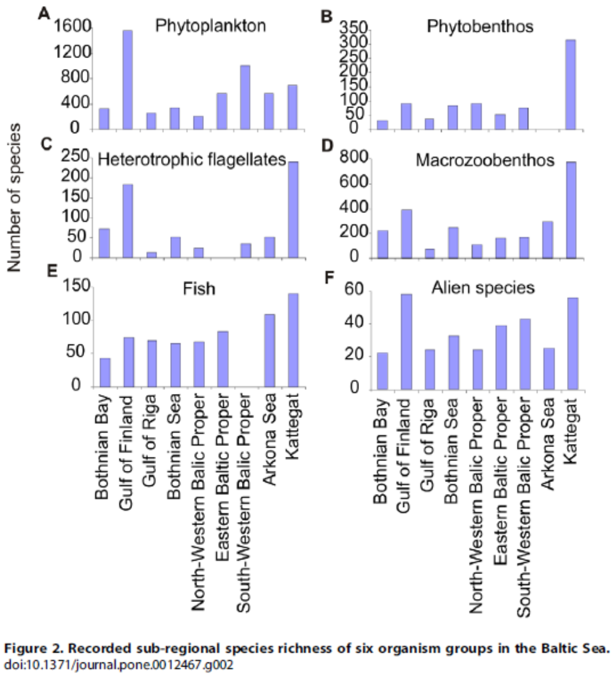

Figure 2.2. Subregional species richness estimated across six groups of organisms (Ojaveer *et al.* 2006).

2.3.2 Greater North Sea, including the Kattegat, and the English Channel

Cluster analyses of survey data for fish, epibenthos and infauna caught in the North Sea were summarised in Rees *et al.* (2007) to highlight the range of assemblages for these groups (Figures 2.3–2.4) and there are clear bathymetric divisions in the various faunal assemblages in the North Sea, broadly equating with different faunas in waters <50 m, 50–100 m and 100–200m. The fauna of the deeper parts of this subregion (e.g. Norwegian Deeps) also have a characteristic and distinctive fauna. The use of such data could usefully inform on appropriate subdivisions in this part of the subregion.

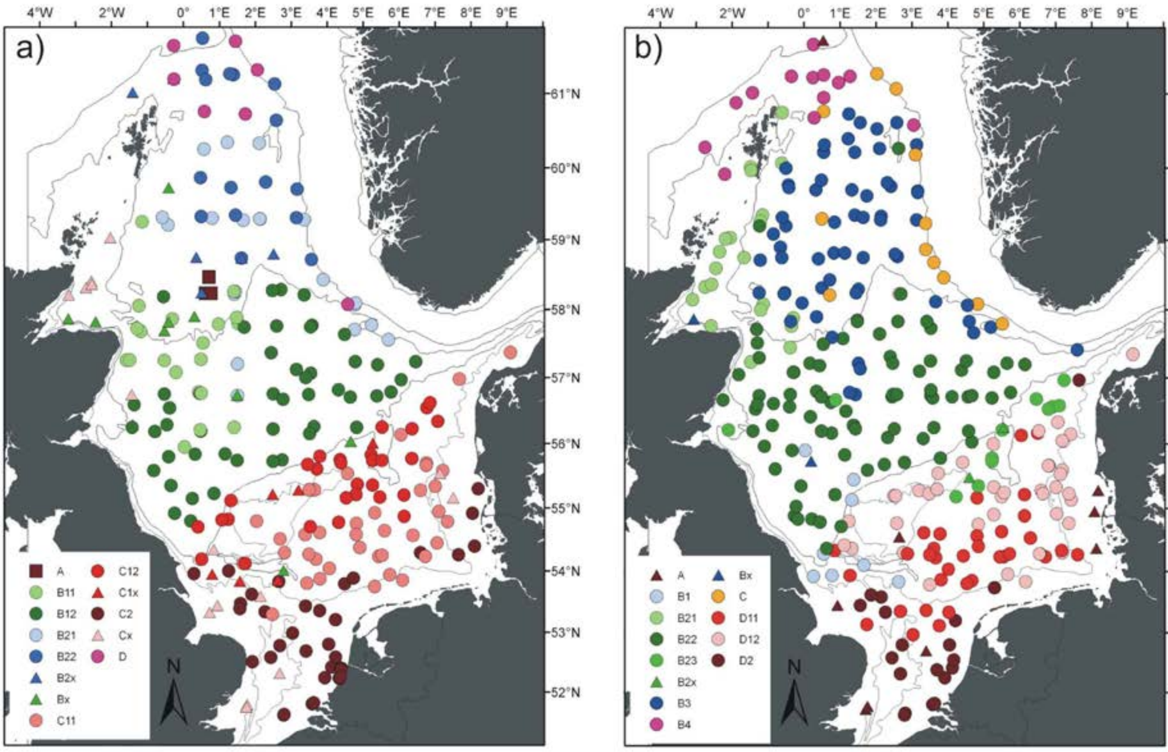

Figure 2.3. Distribution of (a) epifauna and (b) fish assemblages in the North Sea according to the outputs from cluster analyses of fourth-root transformed abundance data (From Rees *et al.*, 2007).

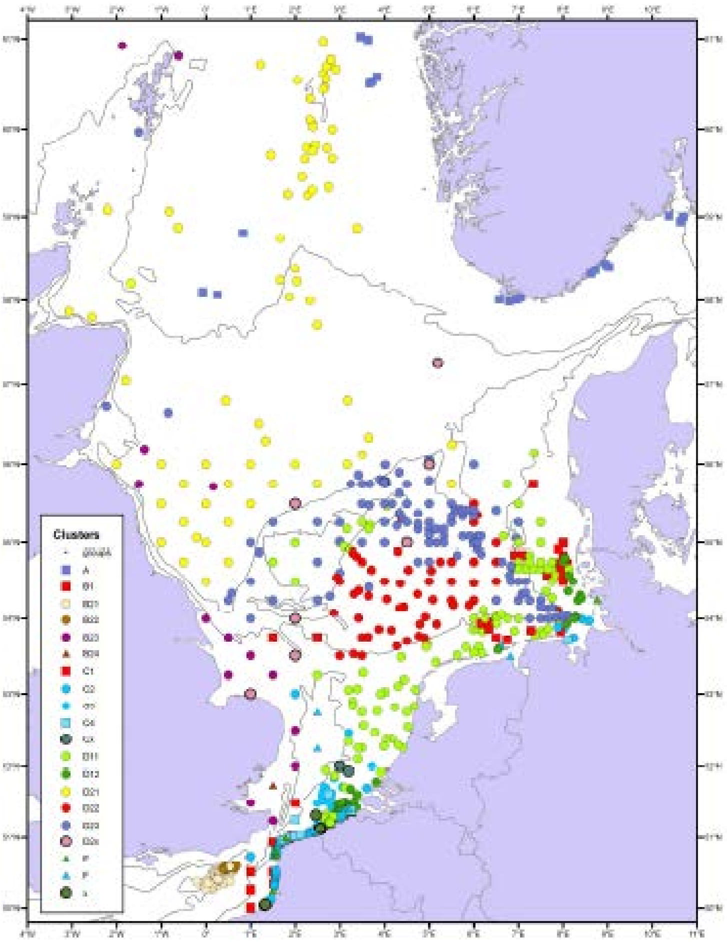

Figure 2.4. Distribution of assemblages in the North Sea in 2000 according to group-average cluster analysis (From Rees *et al.* 2007).

In terms of the English Channel (ICES Divisions VIId-e), there have been many studies of the benthic and demersal assemblages in the area, although many of these have been small-scale studies and there have been fewer studies considering the broad scale area (e.g. Holme 1961; Cabioch 1968; Holme & Wilson 1985; Dewarumez *et al.* 1992; Kaiser *et al.* 1999; Sanvicente-Añorve *et al.* 1996, 2002; Ghertsos *et al.* 2000).

There may be different faunal discontinuities in the English Channel (Holme 1966, Pawson, 1995), broadly equating with the 'narrows' (that part of the English Channel between the Isle of Wight and the Cherbourg Peninsula and the Dover Straits). For example, species such as cuttlefish *Sepia officinalis* and spider crab *Maja brachydactyla* can be widespread and abundant throughout much of the English Channel, but are less numerous in the southern North Sea (i.e. the Dover Straits acts as a faunal boundary). In contrast, some species that are commonly encountered in the parts of the western English Channel, such as anglerfish *Lophius piscatorius,* cuckoo ray *Leucoraja naevus* and sand-star *Astropecten irregularis* are only occasionally found in the eastern English Channel (i.e. the narrows form the faunal boundary). In terms of fish stocks, species such as thornback ray *Raja clavata* in the southern North Sea are considered to extend into the eastern English Channel (i.e. there is a IVc/VIId strock), whereas some of the gadoid stocks in the western English Channel are considered to be a part of wider stocks in the Celtic Sea.

Given these inconsistent faunal boundaries, it is not surprising that there is no consensus in proposed biogeographical boundaries. For example, the whole of the English Channel is included within ICES sub-area VII (i.e. Celtic Seas), whereas OSPAR treats the English Channel within the Greater North Sea and, more recently, ICES suggested that the eastern English Channel and North Sea were one eco-region, with the western English Channel within the Celtic Seas ecoregion. Those species, stocks and habitats to be included within the MSFD that occur in the English Channel ought to be assessed on an appropriate geographical scale.

2.3.3 Celtic Seas

Although there have been few analyses of the benthic or fish assemblages over the entire Celtic Seas regions (partly due to the disparate surveys that have operated in the area, with different types of GOV and other types of trawl used), there is some indication of the types of assemblage that may occur, as indicated below.

There have been few published works on the faunal assemblages in the Hebridean Sea (but see Robertson & Pinn 1999), although there have been several site-specific studies of various lochs.

Demersal assemblages in the Irish Sea have been described (Ellis *et al.*, 2000, 2002a; Ellis & Rogers 2004), and there appear to be distinct inshore assemblages on the sandy environments in shallow coastal waters, with coarser grounds further offshore, and mud banks to the west and south-west of the Isle of Man and off the coast of Cumbria. The shallower waters of the Irish Sea appear to be somewhat different to those of the Bristol Channel (Ellis *et al.*, 2000). This may be due to slightly warmer sea water south of St George's Channel. The benthic communities of St George's Channel and the Bristol Channel have also been described (e.g. Warwick, 1984; Mackie *et al.* 1995, 2006; Wilson *et al.* 2001).

The Celtic Sea has been subject to less investigation (Le Danois 1948; Ellis *et al.*, 2002b), with assemblages varying with sediment and depth. The species occurring in the deeper waters of the Celtic Sea would also appear to be quite similar to those known to occur off North-west Scotland and in the northern North Sea (Ellis *et al.*, 2002b), although more integrated analyses of the various data sets could usefully be undertaken.

The Celtic Sea serves as the northern distribution limit for a variety of Lusitanian species, and some northerly species also have a southern boundary in this area. Some studies have suggested that 49°N may form a more meaningful biogeographical boundary between the typical Celtic Sea fauna and that generally observed further south in the Bay of Biscay (Ellis *et al.*, 2002b; ICES 2005).

2.3.4 Bay of Biscay and the Iberian Coast

In a recent study, Lorance *et al.* (2009) summarized the existing information for several groups of organisms (e.g. microbes, fish, marine mammals), essential habitats (e.g. coastal nurseries), and environmental features (e.g. hydrology, sediments). They highlighted that in the offshore Bay of Biscay, species richness in the macrofauna is dominated by crustaceans, followed by molluscs and echinoderms, while, in the megafauna, molluscs are more numerous. The benthic community of the external shelf margin is dominated **b**y carnivorous polychaetes on sandy-mud shelf bottoms, and by deposit feeders on fine sand bottoms (Le Loc'h *et al.* 2008). There are very few data on the benthic diversity in the Bay of Biscay, and much of the sampling effort and available data has focused on the "Grande vasière". According to Lorance *et al.* (2009) the meiofauna is one of the lesser known benthic components.

Further south, the waters of the Cantabrian Sea and Galician waters have been relatively well studied, and faunal groupings have been described for several areas (e.g. Farina *et al.* 1997; Serrano *et al.* 2006, 2008 and references cited therein). Pascual *et al.* (submitted) and Borja *et al.* (submitted), incorporated zooplankton, benthic flora, benthic fauna, fishes, seabirds and cetaceans into a biodiversity valuation in the waters of the south-eastern part of the Bay of Biscay, in order to assess this qualitative descriptor within the MSFD.

There have also been several studies of Portuguese coastal waters, including coastal areas (e.g. Prista *et al.* 2003; Neves *et al.*, 2008), and some analyses of broad scale data (Gomes *et al.* 2001; Sousa *et al.* 2005, 2006), which could be used to inform on appropriate subdivisions.

2.3.5 Summary

There have been numerous studies on the broad-scale spatial patterns in the structure and composition of various marine groups, including plankton, benthic and epibenthic fauna, and demersal fish. Such studies can usefully inform on the scale and distribution of appropriate subdivisions for monitoring various facets of the marine system under the MSFD. It should also be recognised that, although there boundaries for 'regions' and 'subregions' are defined in the MSFD, such boundaries do not always match with some observed biogeographical boundaries or stock units. When '*relevant species and functional groups*' are identified for assessments under the MSFD, there is a clear role for the relevant ICES Expert Group(s) to comment on the appropriate spatial scale over which they could usefully be assessed.

2.4 Case study 1: Coincident diversity patterns in fish and trawled benthos across the Bay of Biscay coastal nurseries

Coastal and estuarine environments are among the most productive ecosystems in the aquatic environment (Costanza *et al.* 1997). They provide many services to the human population (food, recreational areas etc.) and they play an important role as nursery habitats for many commercial fish and shellfish species. Several studies provided indirect evidence that habitat condition (quality and quantity) prevailing in coastal nurseries can affect the size of some fish populations (Pihl *et al.* 2005).

Using the carbon and nitrogen isotopic compositions of fish and macrobenthic communities, Kopp *et al.* (submitted) studied the spatial distribution of trophic interactions in complex ecosystems such as coastal nurseries. They notably identified several habitats along the estuarine-coastal gradient of the Bay of Vilaine, suggesting the presence of spatial structuring in that ecosystem. A question that can be asked is whether diversity and productivity patterns are also spatialized. To answer this question, the present case study aims to compare spatial distributions of fish and epibenthos in several nurseries across the Bay of Biscay with a view to identifing coincident patterns in species richness (SR) and productivity (in terms of fish density and benthic biomass).

Methods

Ifremer has carried out a number of dedicated coastal nursery surveys along the shore of the Bay of Biscay since the 1980s. The present study focuses on the surveys that have been conducted from 1997–2003 on five nursery grounds located along the French coast of the Bay of Biscay (Figure 2.5). These nursery grounds have been described and classified by Gilliers *et al.* (2006) into 'open shallow muddy estuarine areas under the direct influence of freshwater inflows' (Vilaine, Loire, and Gironde) and 'semi-enclosed sheltered muddy marsh areas with shellfish-farming, little affected by rivers' (Bay of Bourgneuf, Pertuis Antioche).

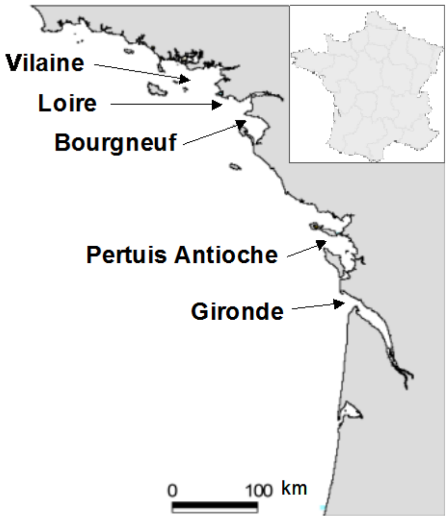

Figure 2.5. Maps of five coastal nurseries along the Bay of Biscay.

The nursery-dedicated surveys were undertaken from the end of August to the end of October. Earlier studies found that this period coincided with the end of the growth phase of juvenile flatfish and that it was a suitable period for their collection, providing consistent estimates for notably 0+ fish age group (Dorel *et al.* 1991). The surveys were conducted using a stratified sampling design according to depth and sediment type. They were carried out in depths ranging from 5–25 m using a 2.9 m wide and 0.5 m high beam trawl with a 20-mm stretched mesh net in the cod-end. Each haul was conducted on homogeneous sediments and depths and lasted 15 min, covering a mean area of 4500–5000 m^2. An average of 22 (± 12 hauls) per year were done in the five coastal areas. All the species caught were counted and the total weight of the haul was recorded.

Species richness was estimated by haul using Margalef's index of species richness (SR):

$$SR = (S - 1)/\ln N$$

where N is the number of individuals (Margalef 1958). This index provides a measure of species richness that is roughly normalized for sample size without using rarefaction techniques. We estimated fish and benthic productivity using fish density (individual/km^2) and benthic biomass (g/km^2). The two indices were standardized by the surface of a haul and the grab, respectively.

Maps of the spatial distribution of the two diversity indices were developed using a systematic grid of 0.03 x 0.03° (Figure 2.6). The mean SR, fish density and benthic biomass were calculated in each cell using observations from the overall time period (1997–2003). For graphical representation, and thus visual comparison, we estimated and represented the quantiles of SR, density and biomass. Therefore, areas of higher SR, densities and biomasses are coloured in red. Statistical comparisons of the spatial distributions of the two diversity indices (SR and biomasses) were tested using

Spearman non-parametric correlations with the two-sided alternative hypothesis. Data analyses and statistical tests were implemented using R software (2008).

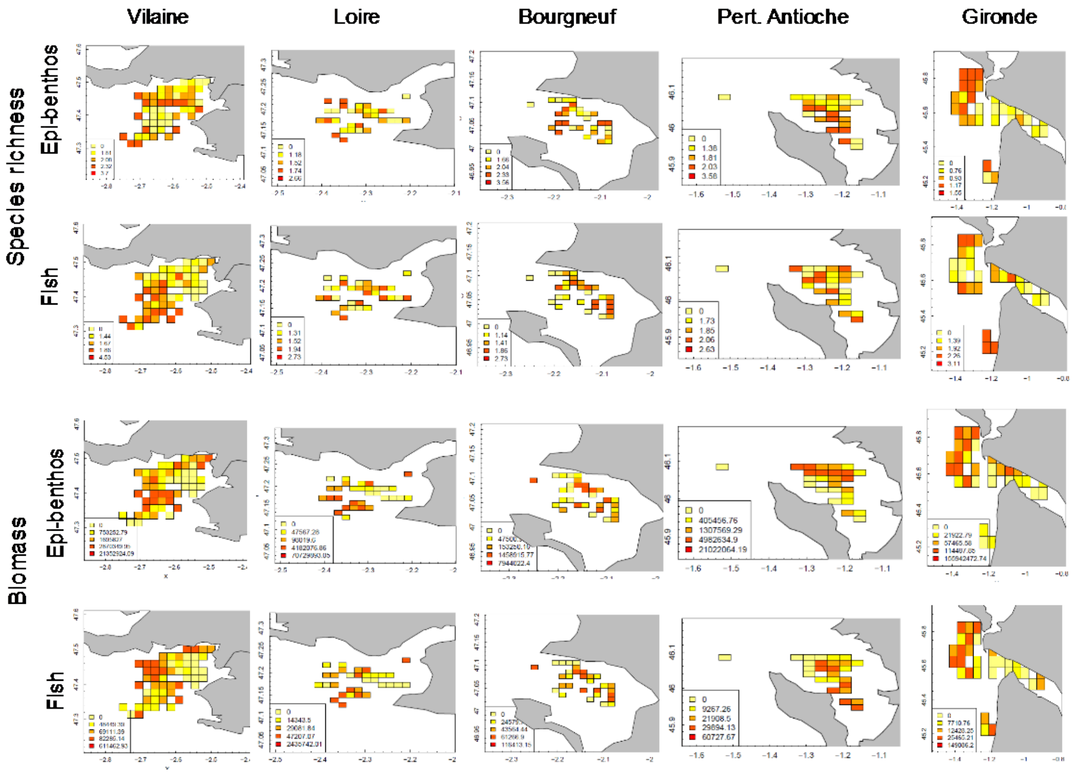

Figure 2.6. Quantile distribution of standardized species richness and biomass calculated for the fish and trawled epibenthos in five coastal nursery grounds of the Bay of Biscay. The colour scale is associated with the quantiles (0%; 0–25%; 25–50%; 50–75%; 75–100%).

Results and Discussion

Comparison of the SR and productivity indices between the five nursery grounds indicated significant concurrent patterns of SR for the Loire (rho = 0.66, p = <0.001) and to a lesser extent the Gironde (rho = 0.34, p = 0.06), whereas the SR patterns in the three other coastal systems were non-significant. Although we did not assess the environmental variability across the five coastal areas, we speculate that the size of the estuary as well as the mouth width may explain these results. Similar results were recently underlined by Nicolas *et al.* (2010) in a study where they analysed the influence of large-scale environmental gradients on estuarine fish species richness from 135 North-eastern Atlantic estuaries from Portugal to Scotland. They observed higher values of species richness in large estuaries and explained their results by the fact that larger estuaries sheltered more diverse habitats and species than smaller ones, and that estuaries with large mouth width are richer in species than both mesohaline and freshwater areas because they offer a greater proportion of areas under high marine influence. Nicolas *et al.* (2010) did not assess the species richness of the epibenthic fauna, but it is very likely that the species richness of the benthos will follow the same broad patterns as for fish species (Durieux *et al.* 2009).

Comparison of the spatial distribution of fish density and epibenthic biomass indicated significant patterns for four out of five nursery grounds, suggesting the presence of spatialized productive areas in almost all the nurseries. Further analyses using environmental variables should be done to identify habitat conditions of high productivity within coastal areas.

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3 Biodiversity indicators

3.1 Introduction

Given the current requirements for EC Member States to monitor 'marine biodiversity' under the Marine Strategy Framework Directive (MSFD), and that various forms of indicator may be required, WGBIODIV proposed the following ToR: "To further explore and assess potential biodiversity indicators, for example by undertaking comparative analyses of taxonomic, functional, surrogate and trophic metrics".

Biodiversity, in its broadest sense, encompasses all of the 'variety of life', encompassing genetic, species and habitat (assemblage and ecosystem) diversity. It is often also presumed to include functional processes, which even further increases the overall scope of 'biodiversity'. Given the complexity and breadth of 'biodiversity', it runs the risk of being a rather abstract concept, which results in questions of whether or not overall biodiversity is even a "measurable entity" (Gaston, 1996a).

Clearly, discrete elements of biodiversity can be measured, for example the genetic diversity of a species or the number of species in a defined area or habitat. Yet, given the range of taxonomic groups, their genetic diversities and roles in the ecosystem, it is clearly not achievable to have a single measure of 'biodiversity' (see Gaston, 1996b and references cited therein).

Species richness has been widely applied as a metric of biodiversity, given that it is often easy to measure and is an understandable measure for the general public. Indeed, some scientists consider this to be the only useful measure of species diversity (Rosenzweig, 1995). It must be recognised that some groups of organisms are better suited to such a measure, either as their taxonomy is better known, that they are sampled effectively in field surveys, or that there is an extensive spatial and/or temporal resolution to existing data. This has resulted in many authors surveying particular groups of species and using sample-richness metrics as proxies for wider 'biodiversity' (Gaston 1996b). It should also be recognised that most marine surveys collect data on a 'gear-specific assemblage' of species, which may not equate with complete taxonomic groups or communities.

There is increasing interest in the functional diversity of ecosystems (see Section 6). Various species are considered to fulfil important ecological functions, thereby providing ecosystem goods and services (e.g. bioturbation, habitat formation, nutrient recycling) in addition to their wider role in food webs (e.g. as top predators or important prey/food resources).

3.2 Metrics of species diversity for faunal assemblages

There are several metrics for examining the species component of biodiversity that are well established in the scientific literature. These include measures of species richness, i.e. the number of species present in a community, assemblage or ecosystem, and evenness, which incorporates information on the abundance of each of the constituent species (the concept of 'species diversity' encompasses the richness and evenness). In practice, these metrics refer to the assemblage of organisms sampled by the survey in question, and not the community *per se*.

There are many benefits of such measures. For example, they are widely used in scientific studies, the concepts can be relatively easily understood by the public, and many existing surveys can be used to generate such metrics.

There are, however, some issues that also need to be considered:

- Some species occur only sporadically in surveys because they are vagrants or extra-limital species. These are natural events, and there is no reason for viewing these species as other than rare occurrences. Vagrancy and range extensions/retractions can be considered important elements of community dynamics and so such species should be considered in diversity metrics.
- Existing surveys will neither sample all the species in a habitat, nor all the species in a taxonomic group. For example, demersal trawl surveys with high headline trawls (e.g. as used in the IBTS) will catch many species of demersal fish and many of the abundant schooling pelagic species, but such trawls will not sample, or not sample effectively, coastal and estuarine species (including diadromous species), reef-associated species, epipelagic species, and large pelagic fish. Survey coverage of habitats off the continental shelf is limited. Comparable issues will also affect existing surveys for other taxonomic groups (e.g. benthos, plankton).
- Trawl surveys often operate over fixed station grids or stratified random sampling (at stations that are known to be 'fishable'), but are based on single samples at each site. The catches at such sites are often viewed as being sufficient to inform on the general composition and structure of fish assemblages, but the lack of replication (or limited number of tows in some assemblages or habitats) may limit accurate monitoring of species diversity metrics. Although benthic surveys generally have some degree of pseudo-replication (e.g. 3–5 grab samples at each station), even this may not fully allow the sampling of all infaunal species.
- Depending on the rationale for the survey and its subsequent survey design, changes in such biodiversity metrics may be more responsive to natural fluctuations in environmental conditions, and so may not be directly attributable to particular human pressures.

- There will be some species of commercial, conservation or ecological interest to managers that might not be taken in such surveys, and so it is possible that basing regional biodiversity monitoring exclusively on such metrics will not inform on 'biodiversity loss'.

Hence, whereas there are benefits of utilising multi-species diversity metrics derived from those surveys that have appropriate spatial and/or temporal coverage in terms of informing on the wider state of the regional assemblages sampled, it will also be important to identify which species, taxa or habitats are not sampled in existing surveys. If any of these are viewed as important species (in terms of potential biodiversity loss), then there is a rationale for developing appropriate species-specific indicators (or identifying what kind of surveys would be required to provide such indicators) to augment 'community' metrics and metrics from other aspects of the ecosystem (e.g. as developed for commercial species).

3.2.1 Monitoring 'biodiversity' versus biodiversity loss

The first qualitative descriptor for determining good environmental status under the MSFD is that "*Biological diversity is maintained…*". The MSFD also states that "*This Directive should also support the strong position taken by the Community, in the context of the Convention on Biological Diversity, on halting biodiversity loss*" (European Commission 2008).

It should be recognised that monitoring 'biodiversity' (e.g. using a community metric from a particular survey) may not necessarily inform on biodiversity loss.

If the various species that are most threatened with local/regional extirpation or extinction are not sampled in a survey, then the survey will only inform on changes in a sub-set of the overall community. It would be possible to present annual metrics of biodiversity that are steady, despite some species being lost. Even if a species is present at the start of a time-series, providing that there are sufficient species that occur sporadically, or appear in later years, then analyses of community metrics may not automatically highlight the loss of a species (although other community analyses may identify the disappearance of a species).

3.3 Identifying (and prioritising) species and habitats for biodiversity monitoring

If it is considered that monitoring 'biodiversity loss' is an important aspect of 'maintaining' biological diversity, then there is merit in identifying those stocks, species, higher taxa and habitats that are most at risk of being lost. If these species are not addressed by indicators for other Descriptors, then there is a rationale for considering to include such species within Descriptor 1 of the MSFD.

In the first instance, those species/habitats that have been identified previously as 'threatened and/or declining' by various fora (e.g. HELCOM, OSPAR, IUCN, CITES or on national wildlife legislation) could be considered. Many of these listings are supported by scientific evidence and/or expert opinion, although the listings of some of these species may be contentious.

Additionally, some of these may be 'flagship species' or 'charismatic megafauna'. Relatively few organisations have attempted to use a standardised approach to assessing the threat status for all the species within a defined group. For example, the IUCN Shark Specialist Group has attempted to undertake assessments for all chondrichthyans in the North-east Atlantic (Gibson *et al.*, 2008). However, the majority of marine groups have not been subject to such a standardised approach, and this may result in some species/stocks that are at risk of being 'lost' not being identified simply by a lack of data collation/analysis, or even though the absence of an 'advocate'.

Additionally, it should be recognised that some of the species listed on regional Red Lists and other conventions (e.g. HELCOM) may include species for which the regional area is outside the main biogeographical range of the species. It could be questioned as to how appropriate it is to list a species as 'threatened' in an area that is only at the fringe of the distribution, particularly in terms of management response.

Some of the European marine species that are listed on various conventions and 'threatened' lists are listed in Table 3.1. In some instances, the data available for some of these species may be too limited to inform on a quantitative indicator.

For example, several species of fish that have been listed as threatened could be considered for use as biodiversity descriptors. However, in some cases, these species may have declined to such an extent that useful metrics may not be realistic. For example, recent scientific trawl surveys have few or no valid records for angel shark and white skate, and there are very few recent data on species such as sawfish in the Mediterranean Sea. Whereas these may represent some of the clearest cases of 'biodiversity loss', the inclusion of such species as indicators (at least in the short term) may not be realistic , and more practical ways of assessing that biodiversity is being maintained are required.

Analyses undertaken by WGECO have helped identify some of the demersal fish species in the North Sea that could be considered as 'vulnerable' (ICES 2009). These included species that had a high biomass and were commercially relevant (in terms of an EU quota existing) but were not assessed. This highlighted species such as common dab *Limanda limanda*, lemon sole *Microstomus kitt*, grey gurnard *Eutrigla gurnardus*, common ling *Molva molva* and starry ray *Amblyraja radiata*.

Other analyses focused on identifying species that were considered vulnerable by combining information on life-history traits, fishing mortality estimates, and population trends. They identified species such as wolf-fish *Anarhichas lupus*, common ling *Molva molva* and blonde ray *Raja brachyura*. In general, the species listed by the latter method typically identified demersal sharks, skates, large gadiforms, anglerfish, anguilliforms and some large flatfish.

It should also be noted that many of these taxa, as well as some members of the Syngnathiformes and Scorpaeniformes, have been identified as vulnerable in other marine ecosystems (See Table 3.1).

In the absence of regional assessments of vulnerability of fish within a region, basing initial lists on those families that are known to be vulnerable in other areas might be useful for informing on the initial selection of candidate species.

Table 3.1. Examples of marine and brackish fish in the ICES area and adjacent waters (including North-west Atlantic) that have been listed as of concern under various nature conservation conventions and wildlife trade organisations. Based on listings of fish included on the UK Wildlife and Countryside Act (WCA), Unites States' Endangered Species Act (ESA), NOAA/NMFS Listings of 'Species of Concern' (SOC), Bern Convention - including Appendices II and III with some species only listed for the Mediterranean region (BC), Canadian Species at Risk Act (SARA), Convention on International Trade in Endangered Species (CITES), or listed as 'Threatened and Declining' (HELCOM, 2007, OSPAR, 2008). This list should not be viewed as complete, and only to indicate the types of Family of fish that have been viewed as of conservation concern in various areas.

FAMILY	SPECIES	LISTING
Petromyzontidae	River lamprey *Lampetra fluviatilis*	HELCOM, BC
	Sea lamprey *Petromyzon marinus*	OSPAR, HELCOM, BC
Squalidae	Spurdog *Squalus acanthias*	OSPAR, HELCOM
Centrophoridae	Gulper shark *Centrophorus granulosus*	OSPAR
	Leafscale gulper shark *Centrophorus squamosus*	OSPAR
Somnosidae	Portuguese dogfish *Centroscymnus coelolepis*	OSPAR
Squatinidae	Angel shark *Squatina squatina*	OSPAR, WCA, BC
Carchariidae	Sand tiger shark *Carcharias taurus*	SOC
Cetorhinidae	Basking shark *Cetorhinus maximus*	OSPAR, WCA, CITES, SOC, BC, SARA
Lamnidae	White shark *Carcharodon carcharias*	CITES, BC
	Shortfin mako *Isurus oxyrinchus*	BC
	Porbeagle *Lamna nasus*	OSPAR, HELCOM, SOC
Scyliorhinidae	Lesser-spotted dogfish *Scyliorhinus canicula*	HELCOM
Carcharhinidae	Dusky shark *Carcharhinus obscurus*	SOC
	Blue shark *Prionace glauca*	BC
Pristidae	Sawfish *Pristis* spp.	CITES
	Smalltooth sawfish *Pristis pectinata*	ESA
	Largetooth sawfish *Pristis perotteti*	SOC
Rajidae	Thorny skate *Amblyraja radiata*	HELCOM, SOC
	Common Skate *Dipturus batis* complex	OSPAR, HELCOM
	Thornback ray *Raja clavata*	OSPAR
	Spotted Ray *Raja montagui*	OSPAR, HELCOM
	White skate *Rostroraja alba*	OSPAR, BC
Mobulidae	Devil ray *Mobula mobular*	BC
Acipenseridae	Baltic sturgeon *Acipenser oxyrinchus*	HELCOM, CITES, SOC
	Sturgeon *Acipenser sturio*	OSPAR, HELCOM,
	Various *Acipenser* spp.	CITES,ESA, SOC
	Adriatic sturgeon *Acipenser naccarii*	BC
	Beluga sturgeon *Huso huso*	BC
Anguillidae	European eel *Anguilla anguilla*	OSPAR, CITES
Clupeidae	Allis shad *Alosa alosa*	OSPAR, HELCOM, WCA, BC
	Twaite shad *Alosa fallax*	HELCOM, WCA, BC
	blueback herring *Alosa aestivalis*	SOC
	Alabama shad *Alosa alabamae*	SOC
	Alewife *Alosa pseudoharengus*	SOC
	Autumn spawning herring *Clupea harengus* subsp.	HELCOM
Cobitidae	Spined loach *Cobitis taenia*	HELCOM
Osmeridae	Rainbow smelt *Osmerus mordax*	SOC
Salmonidae	Houting *Coregonus lavretus oxyrinchus*	OSPAR, WCA
	Whitefish *Coregonus* sp.	HELCOM, BC
	Salmon *Salmo salar*	OSPAR, HELCOM, ESA, BC,

FAMILY	SPECIES	LISTING
		SARA
Gadidae	Cod *Gadus morhua*	OSPAR, HELCOM
	Pollack *Pollachius pollachius*	HELCOM
	Torsk *Brosme brosme*	SOC
Rivulidae	Mangrove rivulus *Rivulus marmoratus*	SOC
Fundulidae	Saltmarsh topminnow *Fundulus jenkinsi*	SOC
Atherinopsidae	Key silverside *Menidia conchorum*	SOC
Trachichthyidae	Orange roughy *Hoplostethus atlanticus*	OSPAR
Syngnathidae	Long-snouted seahorse *Hippocampus guttulatus* (=*H. ramulosus*)	OSPAR, WCA, CITES, BC
	Short-snouted seahorse H*ippocampus hippocampus*	OSPAR, WCA. CITES, BC
	Greater pipefish *Syngnathus acus*	HELCOM
	Black-striped pipefish *Syngnathus abaster*	BC
	Opossum pipefish *Microphis brachyurus lineatus*	SOC
Sebastidae (Scorpaenidae)	Small redfish *Sebastes viviparus*	HELCOM
Cottidae	Miller's thumb *Cottus gobio*	HELCOM
	Fourhorn sculpin *Myoxocephalus quadricornis*	BC
Serranidae	Nassau grouper *Epinephelus striatus*	SOC
	Speckled hind *Epinephelus drummondhayi*	SOC
	Duscky grouper *Epinephelus marginatus*	BC
Sciaenidae	Striped croaker *Bairdiella sanctaeluciae*	SOC
	Brown meagre *Sciæna umbra*	BC
	Shi drum *Umbrina cirrosa*	BC
Stichaeidae	Snake blenny *Lumpenus lampretaeformis*	HELCOM
Anarhichadidae	Atlantic wolf-fish *Anarhichas lupus*	SOC, SARA
	Spotted wolf-fish *Anarhichas minor*	SARA
Gobiidae	Couch's Goby *Gobius couchi*	WCA
	Giant Goby *Gobius cobitis*	WCA
	Canestrini's Goby *Pomatoschistus canestrinii*	BC
	Tortonese's goby *Pomatoschistus tortonesei*	BC
	(A variety of other gobies are also listed under the Bern Convention)	
Scombridae	Bluefin tuna *Thunnus thynnus*	OSPAR
Pleuronectidae	Atlantic halibut *Hippoglossus hippoglossus*	SOC

3.4 Relevant species and groups, potential metrics and indicator development

3.4.1 Background

Appropriate criteria for Descriptor 1 will involve the identification of "*relevant species and functional groups*" (European Commission 2010). The recent JRC/ICES report on Descriptor 1 (see Cochrane *et al.* 2010) has provisionally listed some indicative habitats (Table 3.2) and species groups (Table 3.3).

Table 3.2. Provisional list of predominant habitat types (adapted from Cochrane *et al.*, 2010).

Seabed habitats	Pelagic habitats
Littoral rock and biogenic reef	Low salinity water (Baltic)
Littoral sediment	Reduced salinity water (Baltic, Black Sea)
Shallow sublittoral rock and biogenic reef	Estuarine water
Shallow sublittoral sediment	Coastal water
Shelf sublittoral rock and biogenic reef	Shelf water
Shelf sublittoral sediment	Oceanic water
Bathyal rock and biogenic reef	Ice habitats Ice-associated habitats
Bathyal sediment	
Abyssal rock and biogenic reef	
Abyssal sediment	

Table 3.3. Provisional list of predominant ecotypes for mobile species (adapted from Cochrane *et al.*, 2010).

Higher taxon	Ecotypes
Birds	Offshore surface-feeding birds
	Offshore pelagic-feeding birds
	Inshore surface-feeding birds
	Inshore pelagic-feeding birds
	Intertidal benthic-feeding birds
	Subtidal benthic-feeding birds
	Ice-associated birds
Reptiles	Turtles
Marine mammals	Toothed whales
	Baleen whales
	Seals
	Ice-associated mammals
Fish	Pelagic fish
	Pelagic elasmobranchs
	Demersal fish
	Demersal elasmobranchs
	Deep sea fish
	Deep sea elasmobranchs
	Coastal/anadromous fish
	Ice-associated fish
Cephalopods	Coastal/shelf pelagic cephalopods
	Deep-sea pelagic cephalopods

EC (2010) suggested three criteria for assessing representative species within such groups, reflecting species distribution (distribution range, distribution pattern and area covered by (sessile) species), population size (population abundance and/or biomass) and population condition (demographic characteristics such as the size or age composition, sex ratio, fecundity rates etc.; and population genetic structure).

It should be recognised that the EC Decision Document also refers to Descriptor 1 including information at an ecosystem level, in terms of the "*composition and relative proportions of ecosystems (habitats and species)*" (European Commission 2010). Hence, a combination of multi-species metrics (e.g. from surveys) as well as species-specific metrics will be required to inform on 'biodiversity'.

In terms of the suitability of the species-specific approaches indicated above, this will be heavily dependent on an appropriate selection of species.

3.4.2 Ecotypes

WGBIODIV did not have the time to consider the ecotypes for seabirds and marine mammals. In terms of the suggested ecotypes for fish, there are some potential issues that could also be considered. The RECLAIM project has recently attempted to allocate European fish to ecotypes (see Section 2 of Rijnsdorp *et al.* 2010; Engelhard *et al.* 2011).

Two of the functional groups proposed (coastal/diadromous fish, and ice-associated fish) may to some degree be more ecologically-coherent than the other proposed functional groups of fish. There may be merit in identifying more meaningful assemblages, as opposed to differentiating ecotypes by a taxonomic dichotomy (i.e. elasmobranchs vs. teleosts). It is acknowledged that many elasmobranchs are vulnerable to over-fishing and this group represents some of the clearest cases of regional biodiversity loss. Hence, representative elasmobranchs should be included as species from more biologically meaningful fish assemblages.

Within a region, the spatial distribution of demersal fish assemblages will be related to a combination of factors, including depth, sediment type, water temperature and salinity. Other factors (e.g. sea bed topography, predator-prey interactions) will also operate, but often on smaller spatial scales. There have been several studies illustrating the range of demersal fish assemblages occurring in the ICES area (see Section 11 of ICES 2005 and references cited therein), and so it would be possible for the ICES community to define appropriate demersal fish assemblages for the continental shelf habitats of the ICES area.

In terms of pelagic fish communities, there may be some sense in differentiating the 'small pelagic fishes on the continental shelf' (i.e. herring, sprat, pilchard, anchovy, mackerel and horse mackerel) and the large pelagic fish community (i.e. some of the pelagic sharks, tunas and billfish). These groups, in terms of how they may respond to human activities, may be more meaningful than the suggested taxonomic dichotomy. With regards the deep-water fish assemblages, this too may be a simplistic view of the community, as there are very different assemblages and human activities in different depths.

3.5 Potential for auto-correlation or redundancy in criteria for MSFD monitoring

Given how wide ranging 'biodiversity' is, there is clearly the need for multiple metrics to be developed for use as indicators. Depending on how these constituent metrics are interpreted or aggregated when informing on 'Good Environmental Status', there is the risk that multiple metrics for the different aspects of biodiversity may result in a degree of auto-correlation, and potential redundancy of some indicators.

If one was to consider the biodiversity of fish (Figure 3.1), for example, commercial fish stocks are addressed by Descriptor 3; various fish assemblages (and/or species of interest) may be included in Descriptor 1; some groups of fish are clearly identified in Descriptor 4 (e.g. large fish, short-lived pelagic fish, species at the top of the food chain, anadromous and catadromous migrating species), and there are potential fish-related issues within Descriptor 2, in terms of non-native fish species, which may be more of an issue in the Baltic Sea (e.g. Sapota 2004; Karlson *et al.* 2007).

Species/groups such as small clupeiforms and certain elasmobranchs may therefore be considered important for both Descriptors 3 (commercial species) and Descriptor 4. Hence, there is a rationale for a more integrated approach to developing various 'fish-related' indicators if overlap and redundancy in indicators is to be avoided, and groups working on the various descriptors of Good Environmental Status should not work in isolation.

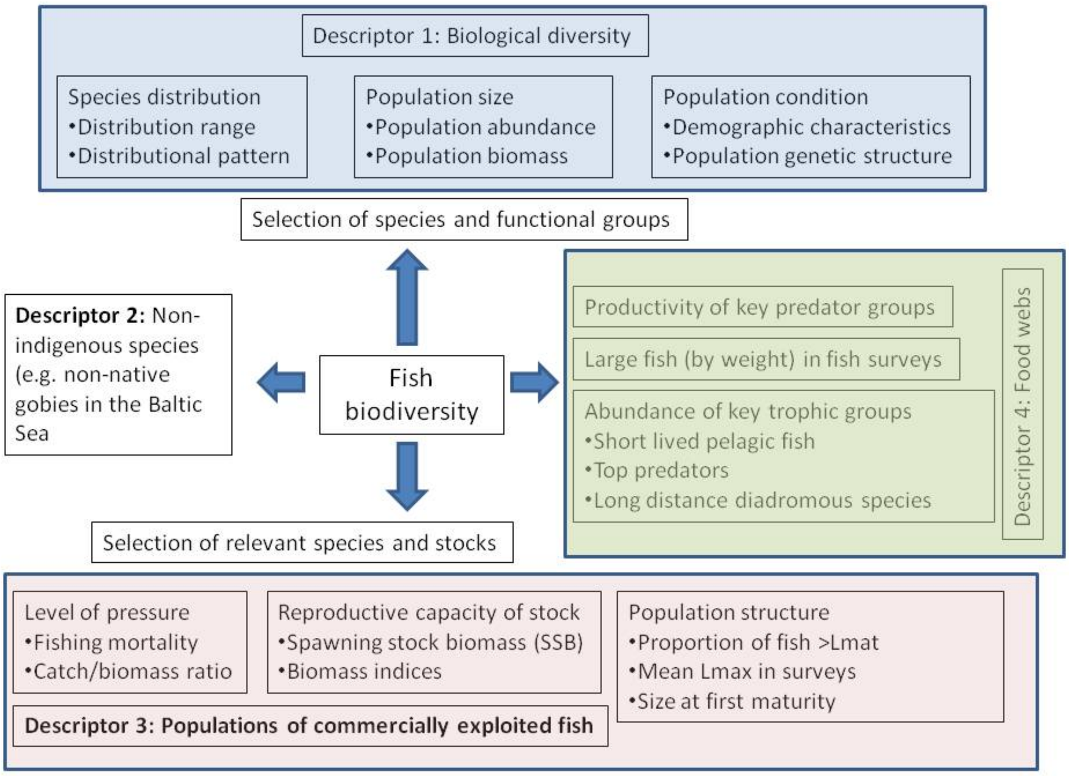

Figure 3.1. Illustration of how some of the various facets of fish 'biodiversity' are addressed by MSFD descriptors.

Such issues are also apparent for other ecological/taxonomic groups, including benthic invertebrates, where benthic communities (Descriptor 1, for species and habitats), non-indigenous species (Descriptor 2), commercial shellfish (Descriptor 3), status of key trophic groups and habitat defining groups (Descriptor 4) and sea-floor integrity (Descriptor 6) may all potentially overlap.

For example, European oyster *Ostrea edulis* is listed as a threatened species by OSPAR and forms habitats (oyster beds) that are also listed by OSPAR. Given that oyster beds can be important for wider biodiversity, are impacted by non-native species in some areas, and can also be affected by toxic algae, issues pertaining to oysters and their beds may be spread over Descriptors 1, 2, 3, 4, 6 (in terms of them being 'habitat-forming groups') and 9.

3.6 Case study of North Sea continental shelf fishes

The demersal fish assemblages of the ICES area (and also large parts of the Mediterranean Sea) are surveyed by internationally-coordinated and other national surveys.

Some of these surveys have operated for many years (e.g. the North Sea IBTS). These surveys provide a valuable data source for examining elements of the biodiversity of the demersal fish assemblage. Those surveys using high headline trawls may also help inform on selected pelagic fish species.

3.6.1 Trends in North Sea demersal fish biodiversity

Perhaps the most elementary indicator of biodiversity is simply an estimate of species richness (S) derived for a specified group of organisms in a specified region using a particular sampling method.

The species richness indicator for the demersal fish assemblage in the North Sea Region derived from the first quarter (Q1) International Bottom Trawl Survey (IBTS) coordinated by ICES suggests an initial decline between 1983 and 1989 followed by an increasing trend up to 2008 (Figure 3.2A). However, there are several issues involved in interpreting such trends, as discussed below.

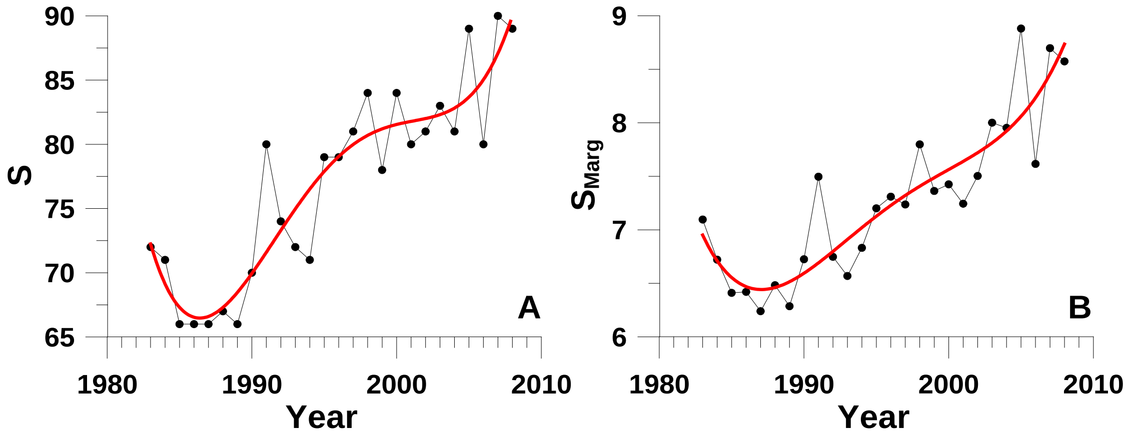

Figure 3.2. Trends in species richness of the North Sea demersal fish assemblage derived from a simple count of the species recorded in the ICES Q1 IBTS each year (A) and estimated using Margalef's index (B), a species richness metric that takes variation in sampling effort into account by adjusting the metric according to the number of individuals sampled in each survey.

3.6.2 Sample size dependency

Species diversity metrics (metrics of species richness and species evenness) are notoriously sample size dependent. The most obvious example of this is provided by species biogeography theory and the species-area relationship: the larger the area of an island the greater the number of species present is likely to be. In terms estimating the species richness of a particular location, for example the number of demersal fish species present in a particular ICES statistical rectangle, this translates as the greater the area within the rectangle that is sampled, the larger the number of species likely to be recorded. Generally this relationship follows one of two non-asymptotic forms, an Arrhenius log-log or Gleason semi-log function, or an asymptotic, Michaelis-Menton function.

Greenstreet & Piet (2008) suggested that a minimum of 20 half-hour IBTS trawls were required to provide estimates of species richness relevant at the ICES statistical rectangle scale. In a more recent study, Greenstreet *et al.* (submitted) argued that sampling effort each year in the Q1 IBTS was sufficient to distinguish inter-annual variation in species richness at the whole North Sea regional scale. That the trend depicted in Figure 3.2A represents real changes in the species richness of the North Sea demersal fish assemblage, rather than an artefact associated with variation in sampling effort, may be inferred from the similar trend shown by Margalef's index of species richness (Figure 3.2B), a metric that scales species richness by the number of individuals included in the sample.

3.6.3 Interpretation of species richness and management objectives

Assuming that the trend in demersal fish species richness in the North Sea shown in Figure 3.2A is real, does this mean that all is well? Species richness is increasing, so is there nothing further to worry about and no need for additional action, just a continuation of the Q1 IBTS survey and regular monitoring of the trend in this metric? The answer to this question depends on what society's goals for North Sea fish biodiversity really are. Despite there being an overall increase in North Sea demersal fish biodiversity, this metric trend could still mask the fact that we may be losing particular species from the assemblage. For example, we may have on-going cases of "biodiversity loss", but if we gain two species for every one that we lose, then this may not be observed with this metric alone.

Defining goals is, therefore, critical. Is the objective simply to maintain a particular level of species biodiversity, or is it that we wish to conserve all components of biodiversity that we currently have? Answering this question may to some extent be dependent on what we judge the causes of this species richness trend to be. It should also be borne in mind that positive trends in species richness, as well as the more commonly associated negative trends, may also be caused by human activities detrimentally affecting the marine ecosystem. Alternatively the observed increase in overall species richness may be the consequence of changes in the marine environment that may be beyond the scope of current marine management regimes to influence directly.

Two recent studies have related changes in the large fish indicator (LFI), the metric that currently supports the OSPAR Ecological Quality Objective for the "Fish Community", to variation in overall indicator of fishing "pressure" on the community. These studies were completely independent of one another, the first analysing the Q1 IBTS data set for the North Sea (Greenstreet *et al.* 2011) and the second analysing the completely separate English Celtic Sea groundfish survey data set (Shepherd *et al.* submitted). Both studies revealed negative responses of the LFI to variation in fishing mortality that involved time-lags of 10 or more years. Adopting an identical approach we examined the relationship between species richness and an indicator of the fishing pressure (community averaged fishing mortality) imposed on the community at various lag periods. At short lags, a significant negative relationship was observed, but as the lag period increased, this relationship became non-significant and then an almost as strong significant positive relationships was detected (Figure 3.3).

Both results can be interpreted as fishing effects. Firstly, the short-term negative relationship could be interpreted as a direct effect of fishing on the abundance of rare species that suffer bycatch fishing mortality. Increasing mortality reduces the abundance of these species to the point where they cease to be sampled effectively by the survey. Reducing fishing mortality has the reverse effect; these rare species populations start to recover and resume their appearance in the survey samples. The long-term positive relationships can be interpreted as an indirect effect of fishing; fishing reduces the abundance of larger piscivorous fish, reducing the natural mortality rates experienced by the rarer (and often smaller) species, so that their populations increase and they start to become more effectively sampled in the survey. Conversely, when fishing mortality decreases, predator populations once again increase and natural mortality rates of the smaller non-target fishes also goes up. Neither explanation invokes any change in the actual species richness of the demersal fish community, just changes in the abundance of the rarest species in the community, affecting their "detectability" by the survey and giving rise to "apparent" changes in species richness.

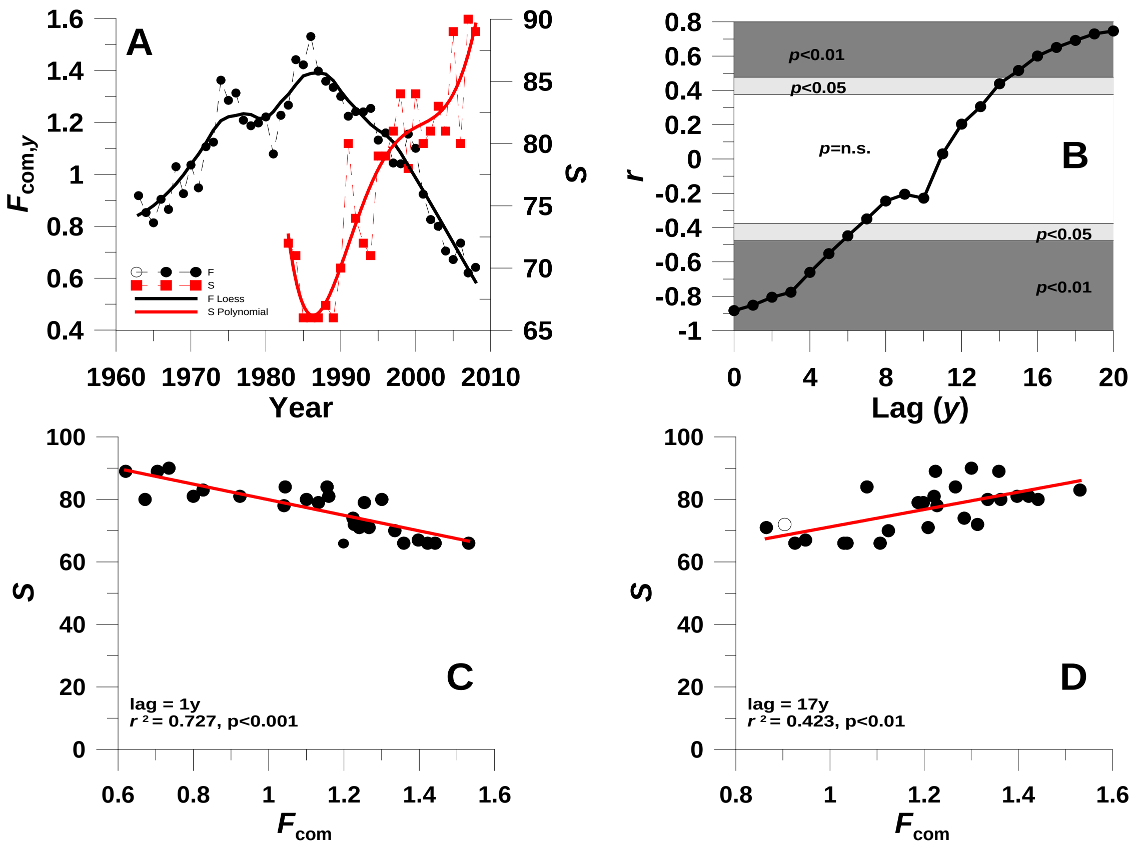

Figure 3.3. A: Trends in North Sea demersal fish species richness (S) and an indicator of community averaged fishing mortality (Fcom). Loess smooths are fitted to the two trend lines. B: variation in the correlation between the two trends at various lags, where the lag represents the period in years (y) separating the date of the species richness observation and the number of years previously that fishing mortality indicator was determined. Negative correlations are shown for illustrative purposes, indicating when the relationship between the two metrics was negative. C: and D: examples of the relationship between species richness and fishing mortality at lags of 1y and 17y respectively. Certainly marked changes in the abundance of nominally 'rare' species have been documented, and these have influenced whether or not these species are detected in the IBTS or not (Harris *et al.*, 2007).

Alternative explanations invoke actual changes in the species richness of the North Sea demersal fish community associated with documented immigrations of new species into the region (e.g. Heessen *et al.* 1996;). Very often such immigrations are linked to a climate change and ocean current change scenarios; for example, a warming North Sea "permits" the invasion and increased abundance of the region by species that normally have a more southerly distribution (Stransky & Ehrich 2001; Beare *et al.* 2004).

One further explanation for the increasing trend in North Sea demersal fish species richness needs to be explored. This is the possibility that the trend is an artefact, driven by increasing taxonomic expertise among the scientists involved in the IBTS, leading to an increased probability of identifying the rare species that get sampled by the survey (Daan 2001, ICES 2007).

To assess the extent to which these various processes may be operating and driving changes in the North Sea demersal species richness indicators, we need to explore the detail contained in the IBTS data set, and we need to examine changes in relative abundance at the species level.

This need for species level analyses is further emphasised by the actual indicators listed for each of the criteria of the Biodiversity Descriptor in the EC Decision document (Table 3.2). Only criterion 1.7.1 for ecosystem structure could perhaps be interpreted as requiring the use of univariate metrics to quantify variation in the relative distribution of individuals between species, size classes, or functional groups within specified species assemblages. Otherwise the stipulated indicators all focus on changes that are apparent within individual species; either changes in abundance, distribution, or demographic characteristics.

For this particular case study we focus on assessing trends in the relative abundance of North Sea demersal fish to attempt to establish a rationale for identifying particular species that could be proposed as candidate indicators for Criterion 1.2.1.

Table 3.2. Levels, criteria and indicator types proposed for Descriptor 1 "Biological diversity is maintained" under the MSFD (Adapted from European Commission 2010).

Level	Criterion	Indicator
Species	1.1 Species distribution	1.1.1 Distributional range
		1.1.2 Distributional pattern within the latter, where appropriate
		1.1.3 Area covered by the species (for sessile/benthic species)
	1.2 Population size	1.2.1 Population abundance and/or biomass, as appropriate
	1.3 Population condition	1.3.1 Population demographic characteristics (e.g. body size or age class structure, sex ratio, fecundity rates, survival/ mortality rates)
		1.3.2 Population genetic structure, where appropriate
Habitat	1.4 Habitat distribution	1.4.1 Distributional range
		1.4.2 Distributional pattern
	1.5 Habitat extent	1.5.1 Habitat area
		1.5.2 Habitat volume, where relevant
	1.6 Habitat condition	1.6.1 Condition of the typical species and communities
		1.6.2 Relative abundance and/or biomass, as appropriate
		1.6.3 Physical, hydrological and chemical conditions
Ecosystem	1.7 Ecosystem structure	1.7.1 Composition and relative proportions of ecosystem components (habitats and species)

3.6.4 Analysis of North Sea Q1 IBTS species abundance trends

The Q1 IBTS time series analysed here spans 26 years from 1983–2008. This is the specific data set used in several recent studies (Greenstreet el al. 2011; Greenstreet *et al.* submitted), which has not yet been updated to include more recent surveys. Using this particular version of the IBTS data allows the results presented here to be compared directly with these previous studies. Following the precedent set by these and even earlier studies, the analyses presented here are restricted to a sub-set of the full database representing the North Sea demersal fish assemblage. In total 128 demersal species have been recorded, of which 49 (38.3%) were recorded in all 26 years (Figure 3.4). Trends in the relative abundance of these 49 species were examined.

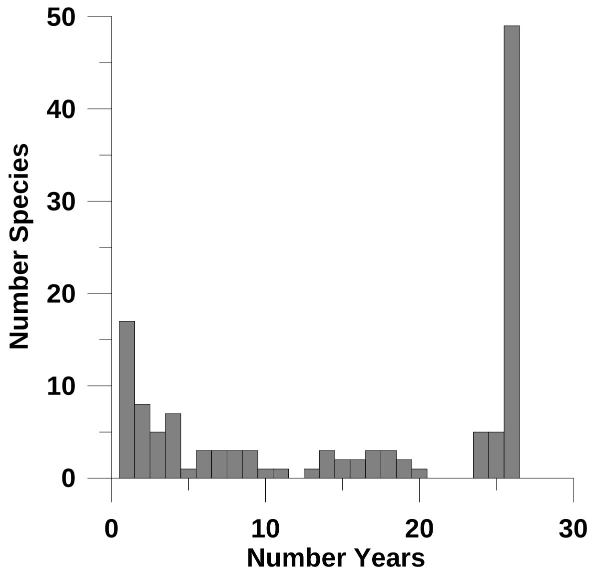

Figure 3.4. Histogram showing the number of demersal fish species recorded a specified number of years in the North Sea Q1 IBTS.

Firstly, the abundance data for each year were standardised to a constant sampled area of 30 km^2 in every year. The actual area sampled in each year varied between 24.2 km^2 and 43.5 km^2. The standardised abundance estimates were then log_{10} transformed. Factor analysis was then performed on these 49 individual abundance trends to assess the level of co-variation between them and to identify the minimum number of independent trend patterns that explained a large proportion of the total variance (Sokal & Rohlf 1981). Varimax orthogonal rotation was applied to minimise the number of variables that had high loadings on each factor. Seven factors each explained >5% of the total variance and combined explained 74.8% of the total variance (Figure 3.5). Additional factors each explained less 4% of total variance and were therefore considered to be non-significant.

Of the seven types of trend identified, only two (factors 2 and 7) indicated recent negative trends, with scores in the last few years of the time series either lower than, or rapidly heading towards, scores observed at the start of the time series (Figure 3.5). Table 3.3 lists the loadings of each species abundance trend on their principal factors. Haddock, torsk and whiting were positively correlated with the scores of Factor 2 and dragonet, hooknose and common dab were positively correlated with the scores of Factor 7. Species abundance trends in these six species do indeed suggest some cause for concern for the three Factor 2 linked species, but the three Factor 7 associated species all currently appear to have a "relatively abundant" status (Figure 3.6).

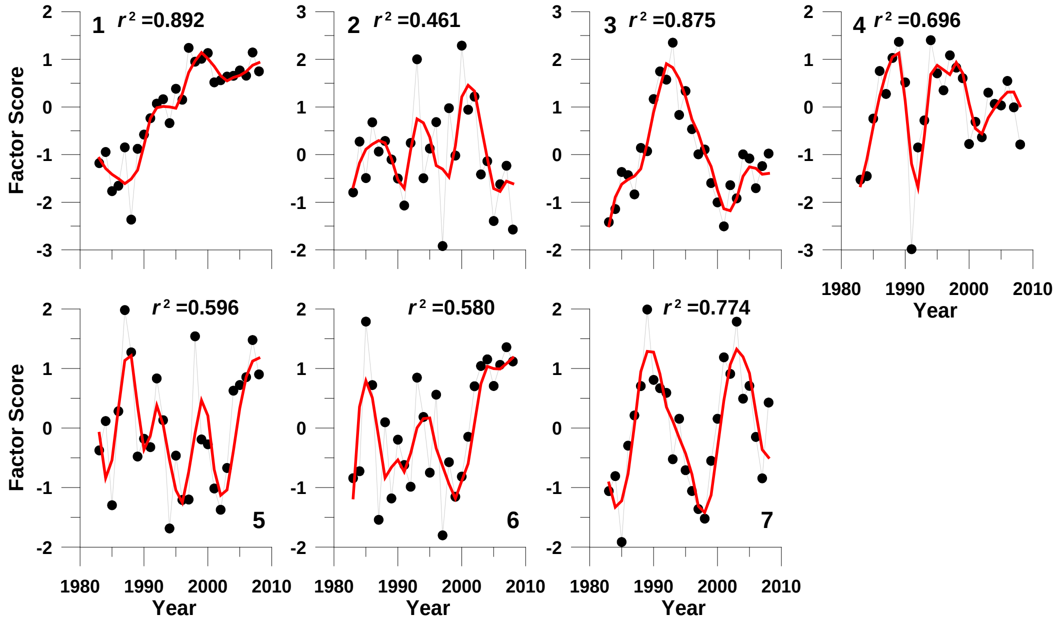

Figure 3.5. Score trends for each of the seven factors explaining >5% of the total variation of a factor analysis applied to annual abundance data for 49 North Sea demersal fish species that were sampled in all of the 26y Q1 IBTS time series. Red curves show loess smoothers derived using a 3rd degree polynomial with tri-cube weighting applied to an 8 data point window.

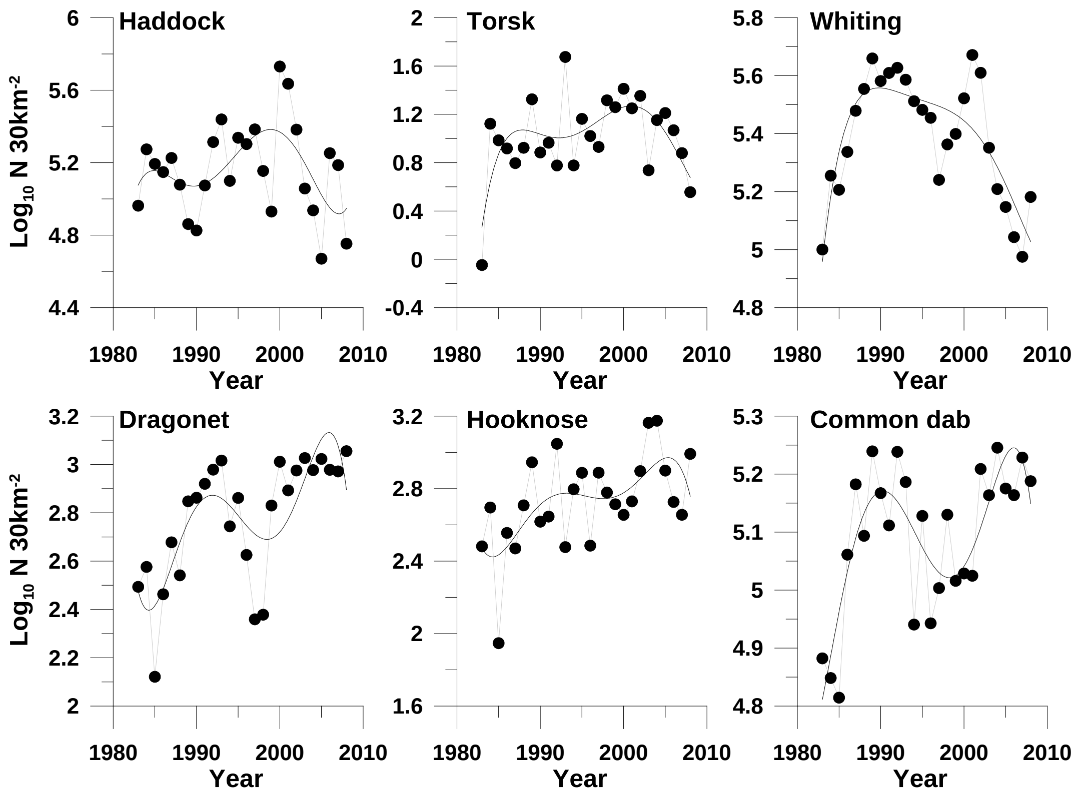

Figure 3.6. Trends in the standardised $\log_{10}$ abundance of the three species, haddock, torsk, and whiting, positively related to Factor 2 and the three species, dragonet, hooknose (pogge), and common dab, positively related to Factor 7. Abundance data are derived from the Q1 IBTS. Smoothers are 5th degree polynomials.

Table 3.3. Loadings of those 49 species recorded in every year (1983–2008)of the Q1 IBTS on each of the seven factors accounting for more than 5% of total variation expressed as the percentage of annual variation in the log_{10} transformed abundance estimates of each species explained by annual variation in the factor scores (r^2). The percentage of the total factor analysis variation explained by each individual factor is given in parenthesis in the factor column header.

Common Name	Specific Name	Factor						
		1 (27.5%)	2 (7.8%)	3 (8.0%)	4 (6.5%)	5 (7.7%)	6 (9.9%)	7 (7.4%)
Grey gurnard	*Eutrigla gurnardus*	0.8440						
Snake blenny	*Lumpenus lampretaeformis*	0.8404						
Long rough dab	*Hippoglossoides platessoides*	0.8268						
Spotted dragonet	*Callionymus maculatus*	0.7937						
Solenette	*Buglossidium luteum*	0.7918						
Scaldfish	*Arnoglossus laterna*	0.7601						
Four-bearded rockling	*Enchelyopus cimbrius*	0.7321						
Lesser weever	*Echiichthys vipera*	0.6562						
Vahl's eelpout	*Lycodes vahlii*	0.5294						
Transparent goby	*Aphia minuta*	0.4858						
Sand goby	*Pomatoschistus minutus*	0.4634						
Lemon sole	*Microstomus kitt*	0.4599						
Lesser-spotted dogfish	*Scyliorhinus canicula*	0.4120						
Five-bearded rockling	*Ciliata mustela*	0.4079						
Brill	*Scophthalmus rhombus*	0.3774						
Anglerfish	*Lophius piscatorius*	0.3514						
Halibut	*Hippoglossus hippoglossus*	0.3108						
Spotted ray	*Raja montagui*	0.2127						
Norway haddock	*Sebastes viviparus*	-0.3008						
Wolf-fish	*Anarhichas lupus*	-0.4583						
Haddock	*Melanogrammus aeglefinus*		0.4197					
Torsk	*Brosme brosme*		0.3372					
Whiting	*Merlangius merlangus*		0.2387					0.2350

Common Name	Specific Name	Factor						
		1 (27.5%)	2 (7.8%)	3 (8.0%)	4 (6.5%)	5 (7.7%)	6 (9.9%)	7 (7.4%)
Spurdog	*Squalus acanthias*		-0.2189					
Sea snail	*Liparis liparis*		-0.2509					
Cuckoo ray	*Leucoraja naevus*			0.5458				
Dover sole	*Solea solea*	0.3617		0.3623				
Flounder	*Platichthys flesus*			0.3612				
Starry ray	*Amblyraja radiata*			0.3230				
Turbot	*Psetta maxima*			0.3079			0.2546	
Plaice	*Pleuronectes platessa*			-0.2939		0.2775		
Lumpsucker	*Cyclopterus lumpus*				0.2778			
Ling	*Molva molva*				-0.1705			
Poor cod	*Trisopterus minutus*				-0.2807			
Thornback ray	*Raja clavata*				-0.4064			
Bib	*Trisopterus luscus*				-0.4272			
Pollack	*Pollachius pollachius*				-0.5652			
Greater weever	*Trachinus draco*					0.6201		
Hake	*Merluccius merluccius*					0.6152		
Bullrout	*Myoxocephalus scorpius*					0.2554		
Saithe	*Pollachius virens*						0.5413	
Tub gurnard	*Chelidonichthys lucerna*						0.4324	
Megrim	*Lepidorhombus whiffiagonis*						0.3097	
Witch	*Glyptocephalus cynoglossus*						-0.2162	
Norway pout	*Trisopterus esmarkii*						-0.2364	-0.2122
Dragonet	*Callionymus lyra*							0.4228
Hooknose	*Agonus cataphractus*							0.3294
Common dab	*Limanda limanda*							0.3210
Cod	*Gadus morhua*							-0.4330

Factor 1 scores showed a relatively consistent increase over the 26 year period covered by our IBTS data set, while trends in the scores of Factors 3, 4, 5, and 6 were characterised by upturns towards the end of the time series (Figure 3.5). Species whose relative abundance trends were negatively related to the scores of these factors might therefore be expected to show a persistent decline in relative abundance over the time period, or more recent down-turns in relative abundance. Table 3.3 identified Norway haddock, wolf-fish (or catfish) (-ve Factor 1), plaice (-ve Factor 3), ling, poor cod, thornback ray, bib, pollack (-ve Factor 4), witch and Norway pout (-ve Factor 6) as fitting this category. The two species negatively related to Factor 1 (Norway Haddock and wolf-fish) appear to have declined markedly over the 26 year period (Figure 3.7). The relative abundance of ling, bib and pollack has also declined (Figure 3.7). However, the relative abundance trends of three other species, plaice, poor cod and thornback ray, do not show cause for concern (Figure 3.7). While the relative abundances of witch and Norway pout are currently at low levels, the relative abundance of these two species appears to have cycled, and the current low levels are no lower than have been observed earlier on in the time series (Figure 3.7).

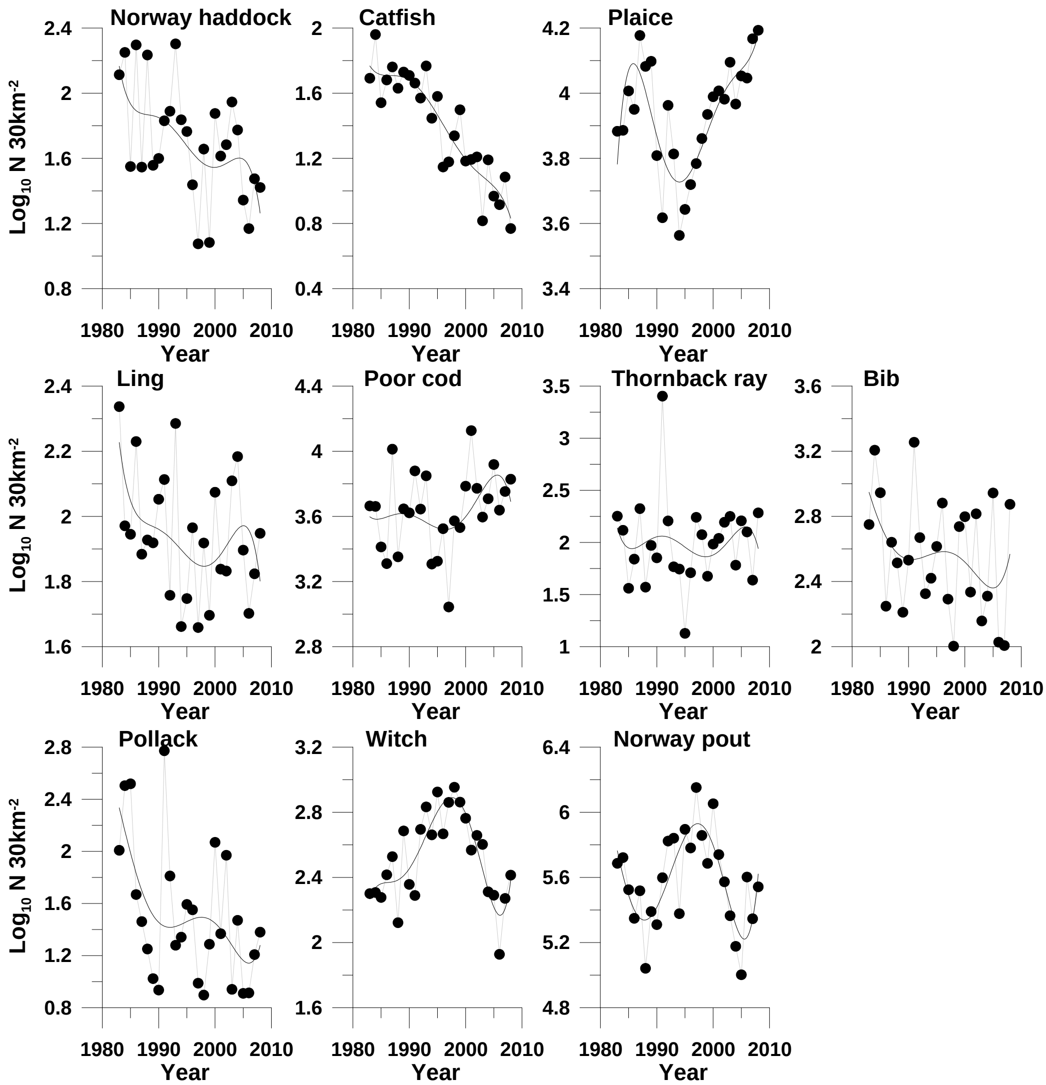

Figure 3.7. Trends in the relative abundance of ten species whose abundance trends are negatively related with trends in the scores of Factors 1, 3, 4, and 6. Smoothers are 5th degree polynomials.

An alternative way to identify potential species-based demersal fish biodiversity indicators is to select species that may be particularly vulnerable to specific human activities and pressures. Piet *et al.* (2009) used a modelling approach to estimate the mortality rate imposed on 39 non-assessed demersal fish species in the North Sea. ICES (2009) applied life-history theory to derive estimates of the level of fishing mortality that each of these species could sustain, and combined these two pieces of information to derive a list of 18 species that they considered were experiencing levels of fishing mortality that were unsustainable (Table 3.4).

Trends in a number of these species have already been considered and the cross-reference to this is given in Table 3.3. Four other species were not recorded in every year of the Q1 IBTS that we have analysed here. Trends in the relative abundance of the remaining nine species have shown either relatively long-term increases, or more short-term increases initiated around 2000 (Figure 3.8).

Table 3.4. Species taken in the North Sea IBTS that have been considered as vulnerable to fishing pressure in the North Sea (Adapted from ICES 2009).

Common name	Scientific name	Assessment results shown/No assessment
Black-mouth dogfish	*Galeus melastomus*	Not continuously represented in IBTS
Smooth hounds	*Mustelus* spp.	Not continuously represented in IBTS
Blonde ray	*Raja brachyura*	Not continuously represented in IBTS
Thornback ray	*Raja clavata*	Figure 6
Spotted ray	*Raja montagui*	Figure 7
Wolf-fish	*Anarhichas lupus*	Figure 6
European eel	*Anguilla anguilla*	Not continuously represented in IBTS
Anglerfish	*Lophius piscatorius*	Figure 7
European hake	*Merluccius merluccius*	Figure 7
Common ling	*Molva molva*	Figure 6
Pollack	*Pollachius pollachius*	Figure 6
Saithe	*Pollachius virens*	Figure 7
Halibut	*Hippoglossus hippoglossus*	Figure 7
Megrim	*Lepidorhombus whiffiagonis*	Figure 7
Flounder	*Platichthys flesus*	Figure 7
Plaice	*Pleuronectes platessa*	Figure 6
Turbot	*Psetta maxima*	Figure 7
Brill	*Scophthalmus rhombus*	Figure 7

Initially this may seem to be counter-intuitive, however, Figure 3.3 indicates that fishing pressure on the demersal community has declined steadily since 1986. With this reduction in fishing pressure, it should not be unexpected that populations of species vulnerable to fishing may well start to recover. Reiss *et al.* (2010) suggested that, in the face of more restrictive TACs, fishermen alter their behaviour so that although the lower TACs brings about a reduction in fishing mortality among target species, this actually has relatively little affect on levels of fishing activity. Explicit in Piet's *et al.* (2009) model is the fact that it is the level of fishing activity that has the greatest influence on the level of mortality experienced by non-target species. Greenstreet *et al.* (2009) presented long-term trends in levels of fishing activity by Scottish fishermen targeting demersal fisheries and showed that levels of demersal fishing activity declined by 40% between 2000 and 2004, driven by a combination of a reduction in fleet capacity, associated with active decommissioning programmes, and the introduction of management measures specifically intended to limit the amount of time that remaining vessels could spend at sea fishing. Since decommissioning and days-at-sea regulations affected the fishing fleets of all EU member states operating in the North Sea, this reduction in fishing activity was not just restricted to the Scottish fleet, demersal fishing activity throughout the North Sea declined by around 25% at this time (Greenstreet *et al.* 2007). The more recent increases in relative abundance of several of the species shown in Figure 3.8 could therefore have been stimulated by this reduction in fishing activity in the North Sea. If we conclude that the recovery in abundance shown by these species is a result of reductions in both overall fishing activity and fishing mortality, then this implies that any reduction in abundance would be indicative of these populations suffering a non-sustainable level of mortality. A target for these particular species abundance indicators could therefore be to avoid negative abundance trends.

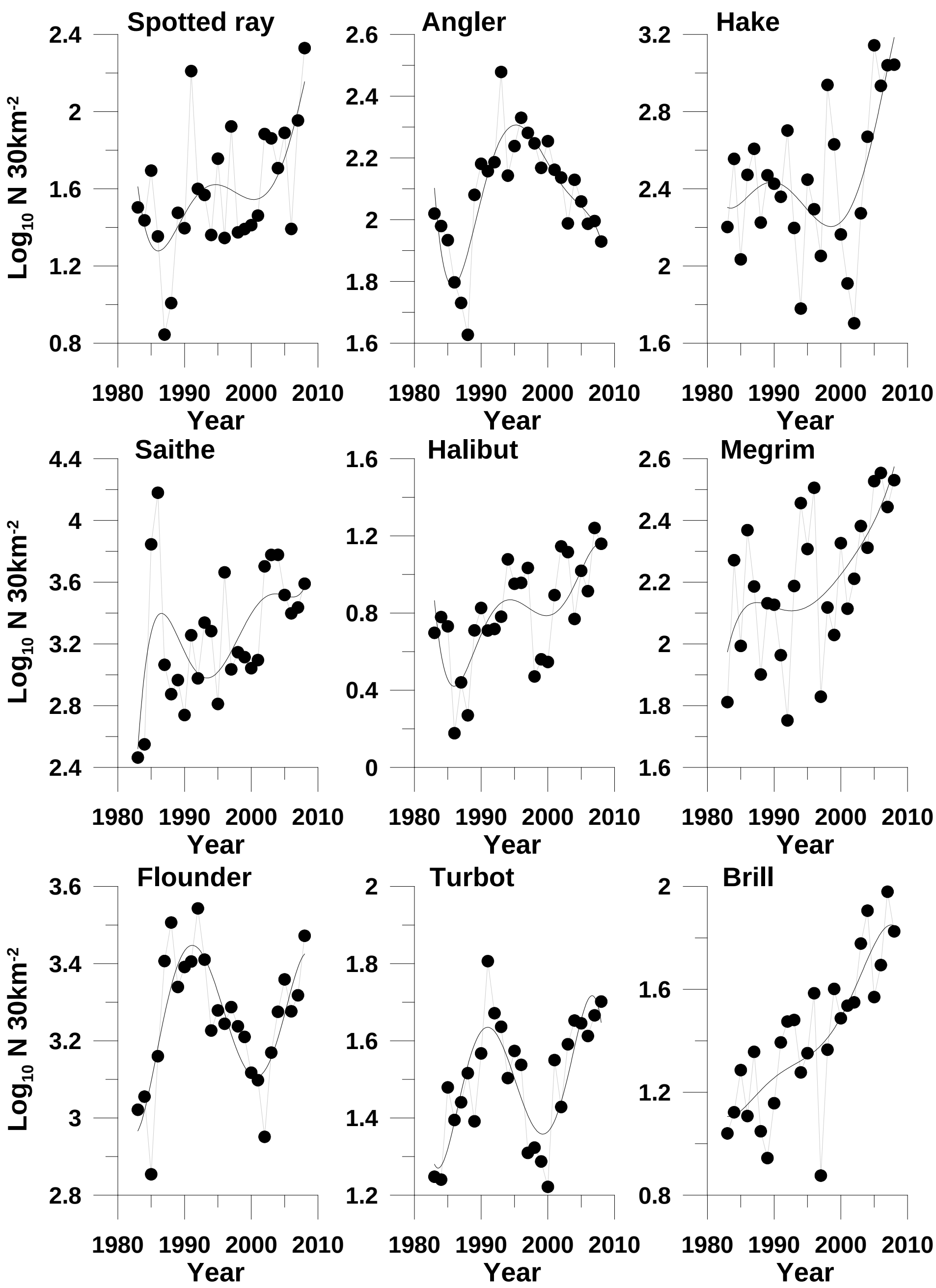

Figure 3.8. Trends in the relative abundance of nine demersal fish species that ICES (2009) considered were likely to have experienced unsustainable levels of fishing mortality.

The trend in anglerfish relative abundance is the clear outlier among the species depicted in Figure 3.8. This species has shown a cyclic trend in abundance with an increase in abundance from the late 1980s to the mid 1990s, followed by a marked and sustained decrease since then. This trend in abundance follows a similar trend in landings of anglerfish from the northern shelf. Variation in this very specific directed fishery bucks the trends in other fisheries, and this may explain difference between the anglerfish abundance trend and the other eight abundance trends shown in Figure 3.8.

3.6.5 Rarely recorded species

Of the 128 species recorded in the whole Q1 IBTS demersal fish database, 17 were only recorded in one year, and a further 20 were recorded in only two, three, or four years. These records may be real, or they could be artefacts resulting from mis-identification of specimens in the field, or species coding errors occurring during the data archiving process. A critical question therefore is to what extent do the records for these 37 species influence the overall trend in the species richness indicator (*S*) shown in Figure 3.2? Figure 3.9A shows the numbers of these 37 species recorded in each year. Subtracting these values from the species richness time series removes the effect of these species on the species richness indicator value in each year. Figure 3.9B shows that the increasing trend in S is robust to this treatment.

Table 3.5 lists the species involved, and most are from families known to be problematic (ICES 2007). Checking the records for these species to assess the likelihood that they are real or false would seem to be a sensible proposition, and could usefully be addressed by IBTSWG. However, whether these records are real or otherwise, the increasing trend in demersal fish species richness is still apparent.

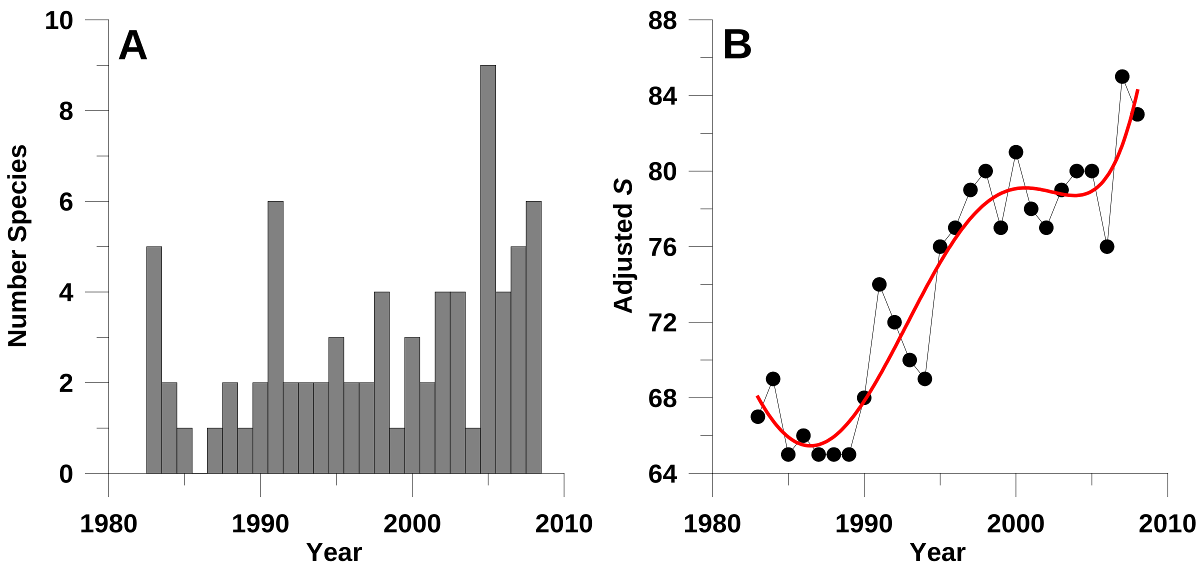

Figure 3.9. A; Histogram showing the number of rarely recorded species (observed in ≤4y) in each year. B: Adjusted North Sea demersal fish species richness indicator excluding these infrequently recorded species.

Table 3.5. List of 37 rarely recorded (in ≤4y) demersal fish species in the North Sea Q1 IBTS, indicating the number of years recorded and stating which years.

Family	Common name	Specific name	Frequency	Years recorded
Petromyzontidae	European river lamprey	*Lampetra fluviatilis*	4	'91,'95,'05,'07
Somnosidae	Greenland shark	*Somniosus microcephalus*	2	'98, '02
Scyliorhinidae	Greater-spotted dogfish	*Scyliorhinus stellaris*	2	'84, '03
Rajidae	Sailray	*Dipturus linteus*	1	'00
	Sandy ray	*Leucoraja circularis*	4	'90, '97, '00, '08
	Speckled skate	*Leucoraja lentiginosa*	1	'96
	Undulate ray	*Raja undulata*	2	'83, '90
Congridae	Conger eel	*Conger conger*	4	'83, '85, '88, '98
Gadidae	Big-eye rockling	*Gaidropsarus macrophthalmus*	1	'05
	Shore rockling	*Gaidropsarus mediterraneus*	2	'88, '91
	Blue ling	*Molva dypterygia*	2	'83, '95
Macrouridae	Murray's rat tail	*Trachyrincus murrayi*	1	'94
Lophiidae	Black bellied angler	*Lophius budegassa*	4	'93, '95, '06, '07
Gobiesocidae	Two-spotted clingfish	*Diplecogaster bimaculata*	3	'87, '91, '92
Syngnathidae	Straight-nosed pipefish	*Nerophis ophidion*	2	'92, '05
Scorpaenidae	Scorpion fish	*Scorpaena scrofa*	1	'91
Triglidae	Streaked gurnard	*Trigloporus lastoviza*	2	'01, '08
Cottidae	Atlantic hook-ear sculpin	*Artediellus europaeus*	1	'97
	Arctic sculpin	*Myoxocephalus scorpioides*	4	'02, '04, '05, '06
	Four-horn sculpin	*Myoxocephalus quadricornis*	1	'03
	Norway bullhead	*Taurulus lilljeborgi*	4	'98, '99, '00, '05
	Ribbed sculpin	*Triglops pingelii*	1	'08
Agonidae	Atlantic poacher	*Leptagonus decagonus*	1	'98
Bramidae	Ray's bream	*Brama brama*	1	'06
Mullidae	Red mullet	*Mullus barbatus*	3	'96, '07, '08
Sparidae	Pandora	*Pagellus erythrinus*	1	'91
Labridae	Small-mouthed wrasse	*Centrolabrus exoletus*	1	'07
	Corkwing wrasse	*Symphodus melops*	3	'05, '06, '07
Stichaeidae	Spotted snake blenny	*Leptoclinus maculatus*	1	'89
Anarhichadidae	Spotted wolf-fish	*Anarhichas minor*	1	'83
Blennidae	Tompot blenny	*Parablennius gattorugine*	1	'05
Gobiidae	Giant goby	*Gobius cobitis*	1	'02
	Common goby	*Pomatoschistus microps*	2	'91, '08
	Painted goby	*Pomatoshistus pictus*	1	'05
Scophalmidae	Eckstrom's topknot	*Zeugopterus regius*	3	'83, '84, '93
Bothidae	Imperial scaldfish	*Arnoglossus imperialis*	4	'94, '01, '03, '08
Soleidae	Sand sole	*Pegusa lascaris*	3	'02, '03, '05

3.6.6 Infrequent species

Forty-two species were recorded in 5–25 years of the 26-year time series (Table 3.6), here referred to as "infrequent species". Six of these species were elasmobranchs, generally considered to be a group particularly vulnerable to over-fishing, and their relative abundance trends are shown in Figure 3.10. Two of these species also featured in the list of species suggested to be vulnerable (Table 3.4; ICES 2009): blonde ray and smooth hound. Data for the two smooth hounds (and it should be recognised that data for these two species are confounded) suggests an increase in relative abundance since the early 1990s, particularly for the starry-smooth hound, whilst blond ray may also have become more abundant in recent years. Again the recovery of these species deemed to be vulnerable to fishing mortality may be attributable to reduced fishing pressure on the demersal fish community. However the relative abundance of the common skate complex has decreased since the start of the IBTS time series, and data for this species suggests little in the way of any recovery in the North Sea. The data for tope and shagreen ray are too sparse to really assess their trends.

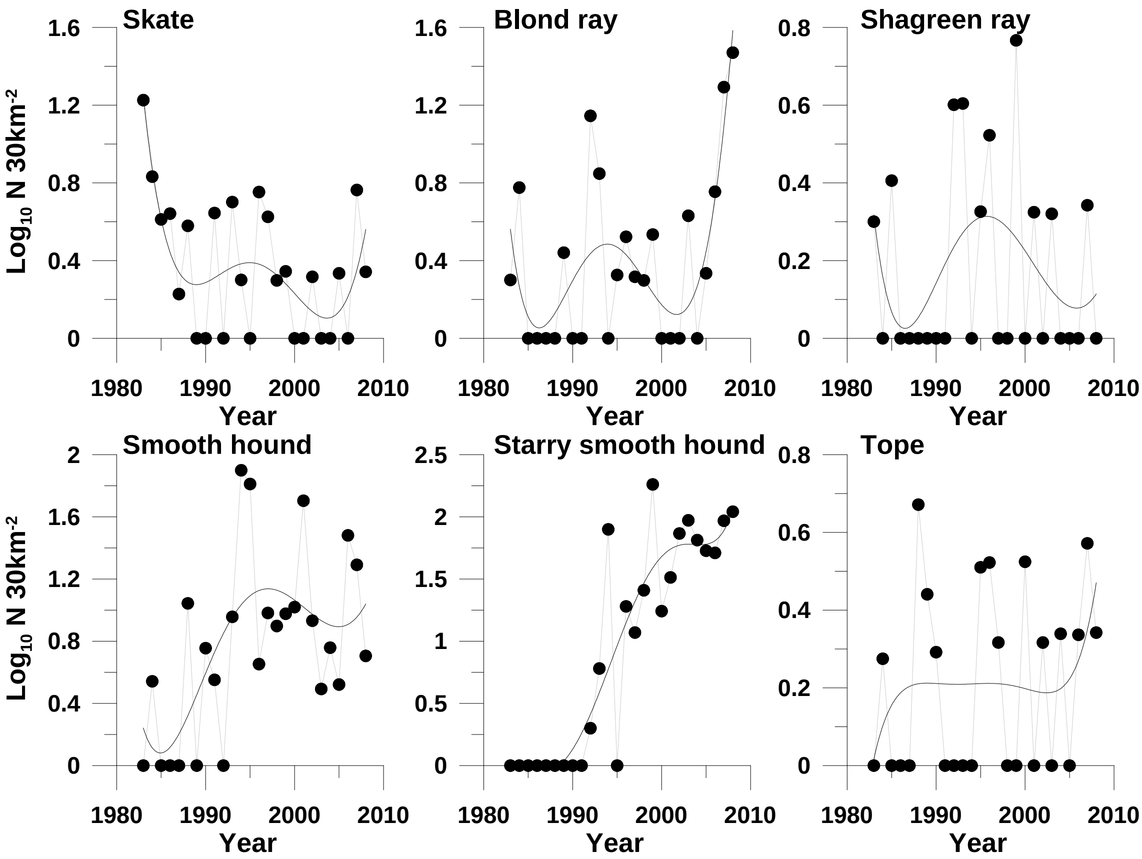

Figure 3.10. Trends in the Log_{10} abundance (+1) per 30 km² area surveyed of six infrequently caught elasmobranch species.

Table 3.6. List of 42 infrequent species in the North Sea demersal fish community (i.e. those recorded in 5–25 years of the 26 year Q1 IBTS).

Common Name	Scientific Name	Number of Years Recorded
Greater pipefish	*Syngnathus acus*	25
Nilsson's pipefish	*Syngnathus rostellatus*	25
Norwegian topknot	*Phrynorhombus norvegicus*	25
Sea scorpion	*Taurulus bubalis*	25
Viviparous blenny	*Zoarces viviparus*	25
Butterfish	*Pholis gunnellus*	24
European eel	*Anguilla Anguilla*	24
Red gurnard	*Chelidonichthys cuculus*	24
Snake pipefish	*Entelurus aequoreus*	24
Three-bearded rockling	*Gaidropsarus vulgaris*	24
Smooth hound	*Mustelus mustelus*	20
Rabbit ratfish	*Chimaera monstrosa*	19
Topknot	*Zeugopterus punctatus*	19
Hagfish	*Myxine glutinosa*	18
Reticulated dragonet	*Callionymus reticulatus*	18
Striped red mullet	*Mullus surmuletus*	18
Sar's eelpout	*Lycenchelys sarsii*	17
Moustache sculpin	*Triglops murrayi*	17
Common skate complex	*Dipturus batis*	17
Starry smooth hound	*Mustelus asterias*	16
Thickback sole	*Microchirus variegatus*	16
Boarfish	*Capros aper*	15
Blonde ray	*Raja brachyura*	15
Bluemouth redfish	*Helicolenus dactylopterus*	14
Montagu's sea snail	*Liparis montagui*	14
Tadpole fish	*Raniceps raninus*	14
Tope	*Galeorhinus galeus*	13
Sea lamprey	*Petromyzon marinus*	12
Pearlfish	*Echiodon drummondii*	11
Fries's goby	*Lesueurigobius friesii*	10
Northern rockling	*Ciliata septentrionalis*	10
Shagreen ray	*Leucoraja fullonica*	10
Black goby	*Gobius niger*	9
Crystal goby	*Crystallogobius linearis*	9
Fifteen-spined stickleback	*Spinachia spinachia*	9
Broad-nosed pipefish	*Syngnathus typhle*	7
Black sea bream	*Spondyliosoma cantharus*	7
Velvet belly	*Etmopterus spinax*	7
Greater forkbeard	*Phycis blennoides*	6
Coldsinny wrasse	*Ctenolabrus rupestris*	6
Redfish	*Sebastes marinus*	6
Ballan wrasse	*Labrus bergylta*	5

Most flatfish species that we have examined so far have shown increasing trends in relative abundance. Two more flatfish species are listed in Table 3.6, topknot and thickback sole, and both have increased in relative abundance since the late 1990s. However, the step function type increase shown by thickback sole may instead be indicative of a species identification issue; recognising this to be a separate species, which previously had been identified as something else (Figure 3.11), and further investigations of these records are required.

Marked invasions into the North Sea by two species, bluemouth and snake pipefish, have been well documented (Heessen *et al.* 1996; Harris *et al.* 2007), and their trends in relative abundance in the Q1 IBTS shown the pattern anticipated for such events (Figure 3.11).

The common eel, another species considered by ICES (2009) to be especially vulnerable to fishing mortality (Table 3.4), was recorded in all but two years of the Q1 IBTS. Although this species has shown an apparent increased in relative abundance in recent years (Figure 3.11), it should be recognised that catch rates are very low and offshore trawl surveys are not the most suitable method for examining the status of this species.

Chimaeras may also be vulnerable to fishing pressure, but trends in the relative abundance of rabbitfish are inconclusive (Figure 3.11).

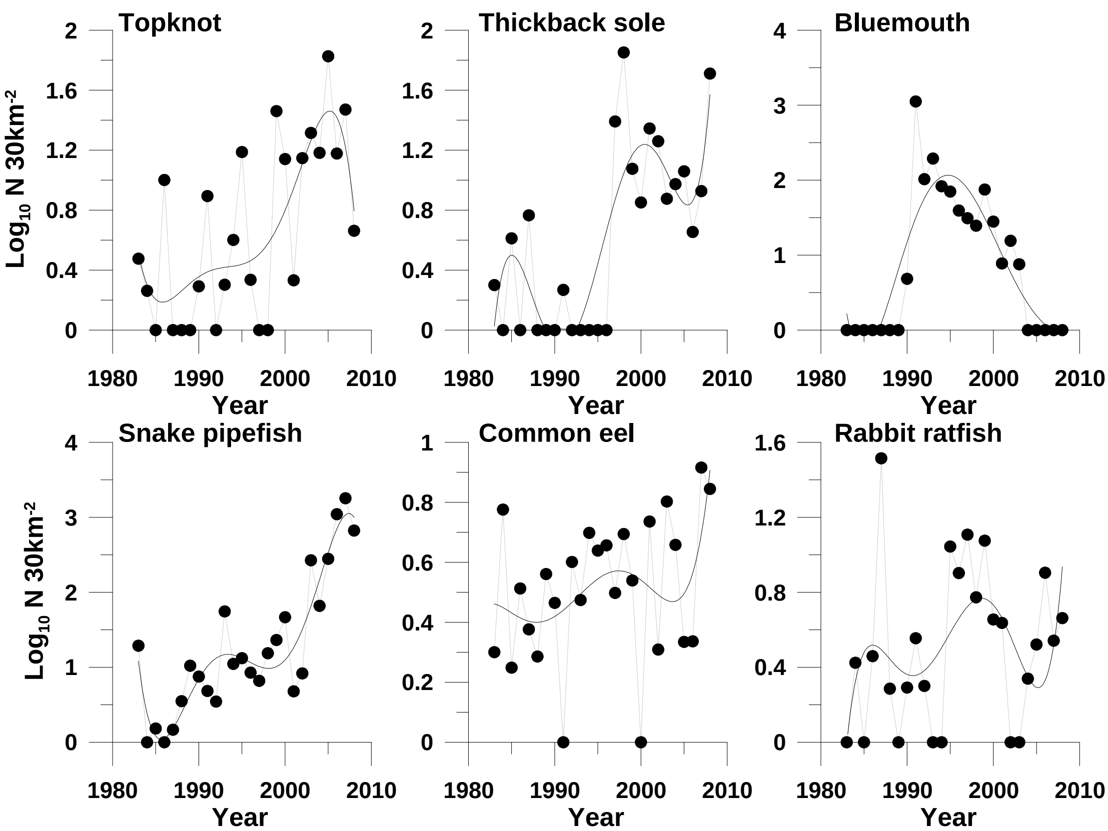

Figure 3.11. Trends in the Log₁₀ abundance (+1) per 30 km² area surveyed of two lesser known flatfish species, two species known to have had marked invasions into the North Sea, a species considered by ICES (2009) to be under threat from fishing mortality, and a further species whose life-history characteristics may also render it particularly vulnerable to fishing pressure.

Ten species were sampled in all but one or two years of the IBTS survey, suggesting that these species were sampled sufficiently frequently that trends in their relative abundance (or frequency of occurrence for species only taken in low numbers) might serve as indicators of the biodiversity of the North Sea demersal fish community. Trends in the relative abundance of two of these (snake pipefish and common eel) were discussed above, and trends in the relative abundance of the remaining eight are shown in Figure 3.12.

Two pipefish species (greater and Nilssens's) and red gurnard have shown marked increases in relative abundance over the course of the Q1 IBTS time series, whilst the relative abundance of butterfish appears to have increased in recent years. Three species, Norwegian topknot, sea scorpion and three-bearded rockling have shown little trends. Only the relative abundance of viviparous blenny seems to have declined.

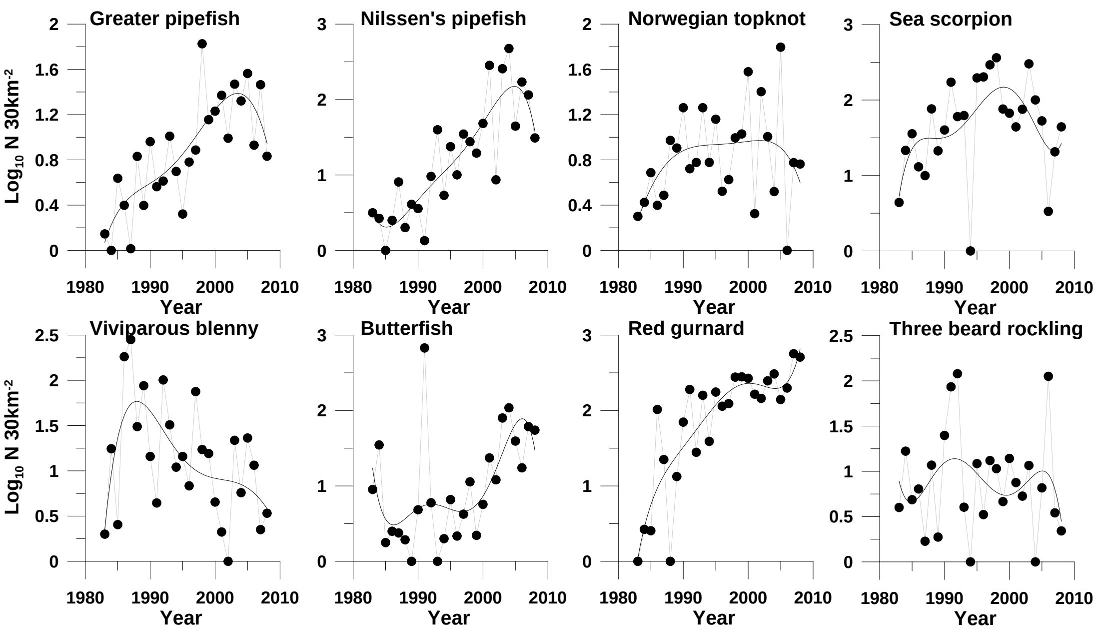

Figure 3.12. Trends in the Log10 abundance (+1) per 30 km² area surveyed of eight species sampled in all but one or two years of the IBTS survey.

So far our analysis has focused on examining the long-term trends in the relative abundance of various species, but the Factor analysis and the large majority of the individual species trends that we have presented here suggest that most species have undergone relatively complex changes in relative abundance over the course of the Q1 IBTS survey period. Faced with such complex trends, identifying species whose potential abundance trend could be used to "populate" indicator 1.2.1 of the MSFD, and then setting a target for such an indicator is a difficult task.

Perhaps a simpler approach would be to consider the state of each species population size over the last four years of the time series compared with the average population condition over the entire time series. To illustrate this approach we considered only those species recorded in at least half of the years (13 years) when the survey took place; a total of 76 species. Table 3.7 gives the average abundance deviation from the mean population abundance over the whole 26y period of the IBTS survey with the deviation expressed as standard deviation units, thus standardising the data for all species to allow direct comparison. Standardisation followed the procedure:

$$D = \frac{x - \bar{x}}{\sigma},$$

where D is the deviation metric, x is the annual abundance; $\bar{x}$ is the mean abundance over the whole time series and σ is the standard deviation. The metric shown in Table 3.7 is the average D computed over the last four years of the time series. Of the 76 species where data were considered adequate for this analysis, 13 species showed negative abundance deviations at the end of the time series of >0.5 SD of which, four species (whiting, wolf-fish, lumpsucker and witch) showed negative deviations of >1SD. This compared with a total of 40 species that showed positive abundance deviations of >0.5 SD, of which 17 were >1 SD.

Table 3.7. List of 76 species giving the average population abundance deviation (expressed in standard deviation units) for the period 2005 to 2008 relative to the mean population abundance between 1983 and 2008. Cell colour coding is: green >+1SD; pale green >+0.5SD to +1SD; red <-1SD; pale red <-0.5SD to -1SD.

Scientific Name	Common Name	Deviation (SDs)
Scyliorhinus canicula	Lesser-spotted dogfish	1.471
Mustelus mustelus	Smooth-hound	0.406
Mustelus asterias	Starry smooth-hound	1.043
Galeorhinus galeus	Tope	0.443
Chimaera monstrosa	Rabbitfish	0.375
Leucoraja naevus	Cuckoo ray	-0.240
Raja brachyura	Blonde ray	1.315
Dipturus batis	Common skate	-0.020
Raja montagui	Spotted ray	0.925
Amblyraja radiata	Starry ray	-0.898
Raja clavata	Thornback ray	0.149
Anguilla anguilla	European eel	0.409
Trisopterus luscus	Bib	-0.268
Gadus morhua	Cod	-0.900
Ciliata mustela	Five-bearded rockling	1.144
Enchelyopus cimbrius	Four-bearded rockling	0.188
Melanogrammus aeglefinus	Haddock	-0.774
Molva molva	Ling	-0.522
Pollachius pollachius	Pollack	-0.755
Trisopterus esmarkii	Norway pout	-0.703
Trisopterus minutus	Poor cod	0.617
Pollachius virens	Saithe	0.546
Gaidropsarus vulgaris	Three-bearded rockling	0.104
Raniceps raninus	Tadpole fish	-0.245
Brosme brosme	Torsk	-0.263
Merlangius merlangus	Whiting	-1.419
Merluccius merluccius	Hake	1.560
Anarhichas lupus	Wolf-fish	-1.394
Callionymus lyra	Dragonet	0.897
Callionymus reticulatus	Reticulated dragonet	1.141
Callionymus maculatus	Spotted dragonet	0.677
Pomatoschistus minutus	Sand goby	0.792
Aphia minuta	Transparent goby	0.806
Mullus surmuletus	Striped red mullet	1.138
Pholis gunnellus	Butterfish	0.875
Sebastes viviparus	Norway haddock	-0.999
Lumpenus lampretaeformis	Snake blenny	0.659
Trachinus draco	Greater weever	0.424
Echiichthys vipera	Lesser weever	0.916
Lycenchelys sarsii	Sar's eelpout	0.663
Zoarces viviparus	Viviparous blenny	-0.460

Scientific Name	Common Name	Deviation (SDs)
Lycodes vahlii	Vahl's eelpout	0.273
Syngnathus acus	Great pipefish	0.694
Syngnathus rostellatus	Nilsson's pipefish	0.862
Entelurus aequoreus	Snake pipefish	1.787
Lophius piscatorius	Anglerfish	-0.492
Myxine glutinosa	Hagfish	0.877
Agonus cataphractus	Hooknose (pogge)	0.336
Myoxocephalus scorpius	Bullrout	1.023
Triglops murrayi	Moustache sculpin	0.454
Taurulus bubalis	Sea scorpion	-0.552
Cyclopterus lumpus	Lumpsucker	-1.060
Liparis montagui	Montagu's sea snail	0.596
Liparis liparis	Sea snail	0.550
Helicolenus dactylopterus	Blue-mouth redfish	-0.928
Eutrigla gurnardus	Grey gurnard	1.128
Chelidonichthys cuculus	Red gurnard	0.800
Chelidonichthys lucerna	Tub gurnard	1.093
Squalus acanthias	Spurdog	0.059
Capros aper	Boarfish	0.408
Arnoglossus laterna	Scaldfish	1.159
Limanda limanda	Common dab	0.759
Platichthys flesus	Flounder	0.602
Hippoglossus hippoglossus	Halibut	1.053
Hippoglossoides platessoides	Long rough dab	0.548
Microstomus kitt	Lemon sole	0.241
Pleuronectes platessa	Plaice	1.056
Glyptocephalus cynoglossus	Witch	-1.103
Scophthalmus rhombus	Brill	1.200
Lepidorhombus whiffiagoni	Megrim	1.305
Phrynorhombus norvegicus	Norwegian topknot	0.019
Zeugopterus punctatus	Topknot	1.092
Psetta maxima	Turbot	0.976
Solea solea	Dover sole	0.442
Buglossidium luteum	Solenette	0.938
Microchirus variegatus	Thickback sole	0.789

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4 Spatial approaches in assessing biodiversity status

4.1 Introduction

WGBIODIV discussed spatial elements to biodiversity at a previous meeting (ICES, 2010), including how spatial methods could be used to assess biodiversity status, the debate over biodiversity 'hotspots', and how the various spatial and temporal patterns in 'diversity' that can occur may directly or indirectly influence survey data.

Hence, WGBIODIV proposed the ToR "To review the existing spatial approaches in assessing biodiversity status, and the spatial and temporal scales on which different elements of marine biodiversity operate, with regards the implications for survey design and indicator development".

Mapping and spatial modelling of biodiversity will help to understand the appropriate scales for specific survey designs. For the planning of station grids, it is essential to know about the distribution of the target species, assemblages and habitats. Fisheries surveys have traditionally been designed to cover the distribution range of the stock(s) of interest; therefore a sound knowledge about fish distributions exists in many regions. For other taxonomic groups, such as benthos, field survey and sampling designs are often limited in space and/or time, especially in offshore waters. Such data gaps hamper robust spatial assessments of biodiversity.

There are comparatively few studies on marine biodiversity 'hotspots' (e.g. Attrill *et al.*, 1996; Bograd *et al.*, 2006). The identification of potential biodiversity 'hotspots' is a contributor to spatial planning and will be demonstrated in Section 4.2, through an example of on-going biodiversity assessments of the Dutch exclusive economic zone.

Section 4.3 addresses some of the caveats when sampling biodiversity across spatial scales; and Section 4.4 discusses spatial patterns of biodiversity in the context of the Marine Strategy Framework Directive (MSFD).

4.2 Spatial approaches to assessing biodiversity status

4.2.1 Mapping and spatial modelling

When assessing biodiversity across regions, mapping is an extremely informative and user-friendly method to visualize the observed patterns. For the sake of simplicity, the sampling of communities or assemblages is often treated as point-abundance sampling, even though sampling of microbes, benthos, epifauna, macrophytes, plankton and fish is integrated differently over areas and depths. When mapping point-abundance data, one is confronted with interpolating abundance estimates between the sampled points. Spatial statistics provide solutions to this problem, e.g. by means of kriging. Many GIS software packages include kriging procedures and have become a well known practice in marine ecology (see Figure 4.1 below, but also Mello and Rose 2005; Stelzenmüller, Ehrich *et al.* 2005; HELCOM 2010).

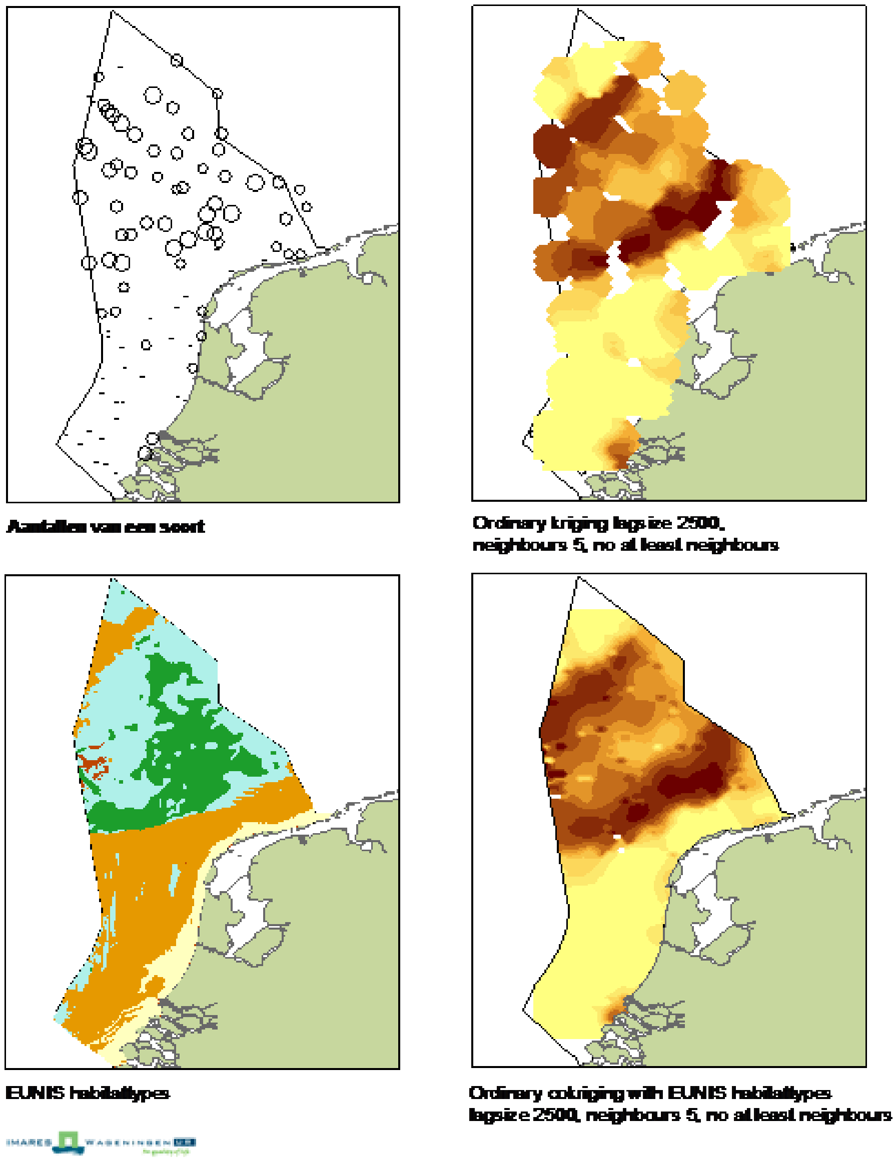

Figure 4.1. Example of spatially extrapolating data using co-kriging with EUNIS habitat types in GIS within the Dutch exclusive economic zone (EEZ). The point-abundance estimates of the bivalve *Abra alba* (upper left) are extrapolated to neighbouring sampling stations or to 0 by ordinary kriging (upper right). The map of habitat types (lower left) is then used to adjust the kriging estimates by accounting for habitat suitability (lower right) (Lindeboom *et al.* 2008).

While the general approach to mapping is to facilitate the visualisation of habitat suitability for species or communities, recent research has focussed on the mapping of biodiversity, rather than habitat suitability. Mapping diversity allows diversity hotspots to be identified in a certain region, based on single point sampling. Techniques for mapping nematode diversity in the Southern Bight of the North Sea (Figure 4.2) were developed by Merckx *et al.* (2010). Species richness and ES(25) were predicted by different methods: ordinary kriging (OK)and regression kriging (RK) with ordinary least squares (OLS) and generalised least squares (GLS). The predictive value of these methods was evaluated by an independent test set. The results indicated that GLS improved the OK models substantially, while RK only slightly improved the GLS model. This technique also allows the environmental variables that are important for generating the diversity maps to be identified. This suggests that the growing need for detailed maps of biodiversity hotspots can be successfully fulfilled by regression and interpolation techniques.

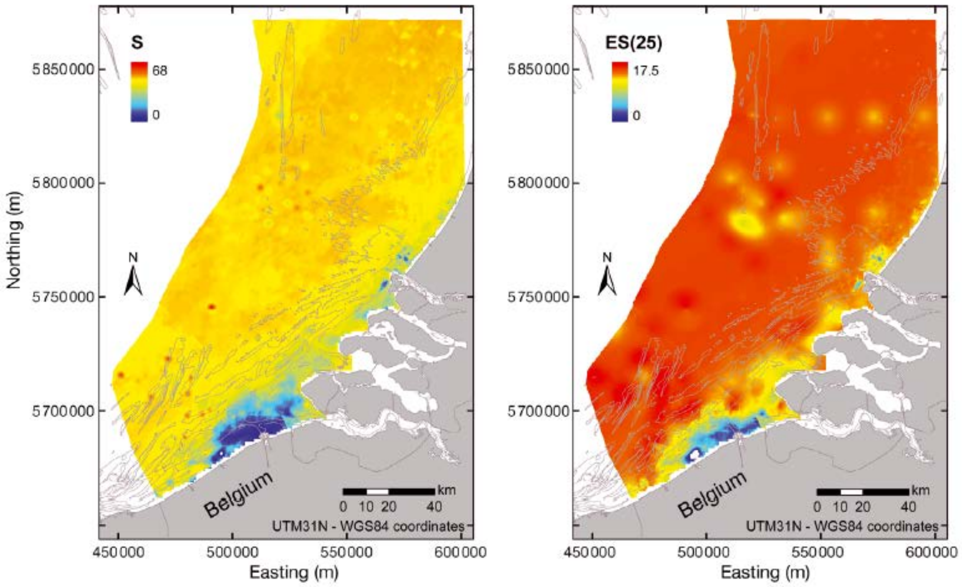

Figure 4.2. Maps of generalised least square models, predicting nematode diversity after kriging of diversity indices S (left) and ES(25) (right). Gray lines in water are bathymetric lines (from Merckx *et al.* 2010).

4.2.2 Spatial aggregation of biodiversity metrics

To combine maps of different biodiversity metrics is not straightforward. Derous *et al.* (2007), as part of the Marbef project, introduced a concept of integrated biodiversity value maps that show the relative value of subareas, based on judgment of biodiversity components (Case study 1). This method requires a lot of system knowledge and is not based on species-specific metrics.

A more direct way of combing maps of biodiversity metrics (e.g. univariate metrics such as species richness and evenness, or species specific metrics) is scaling different maps of biodiversity indicators using GIS (Case study 2) and combining them into a single map. As discussed in Section 2, the question is how the different biodiversity indicators and their maps should be weighted. The question is also on what taxonomic level, temporal and spatial scales can the data be aggregated.

Case study 1: Biodiversity value maps

A region is judged on a set of criteria that compile and summarize all available biological and ecological information. The main criteria are rarity (how common is the area), aggregation (feeding area, etc.) and fitness consequences (spawning area) and the adjusting criteria are naturalness (human impacted) and proportional importance (relative to larger areas). The value is estimated according to the scheme in Figure 4.3 and may deliver maps as is shown in Figure 4.4. This approach has been used for several coastal areas across Europe (Belgium, UK, Poland, Portugal), and most recently along the Basque coast (Bay of Biscay) (Pascual *et al.*, submitted).

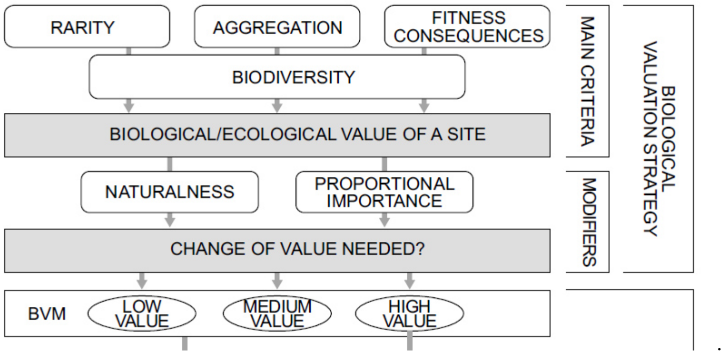

Figure 4.3. Scheme (partly shown) to assess the biodiversity value of an area (From Derous *et al.* 2007).

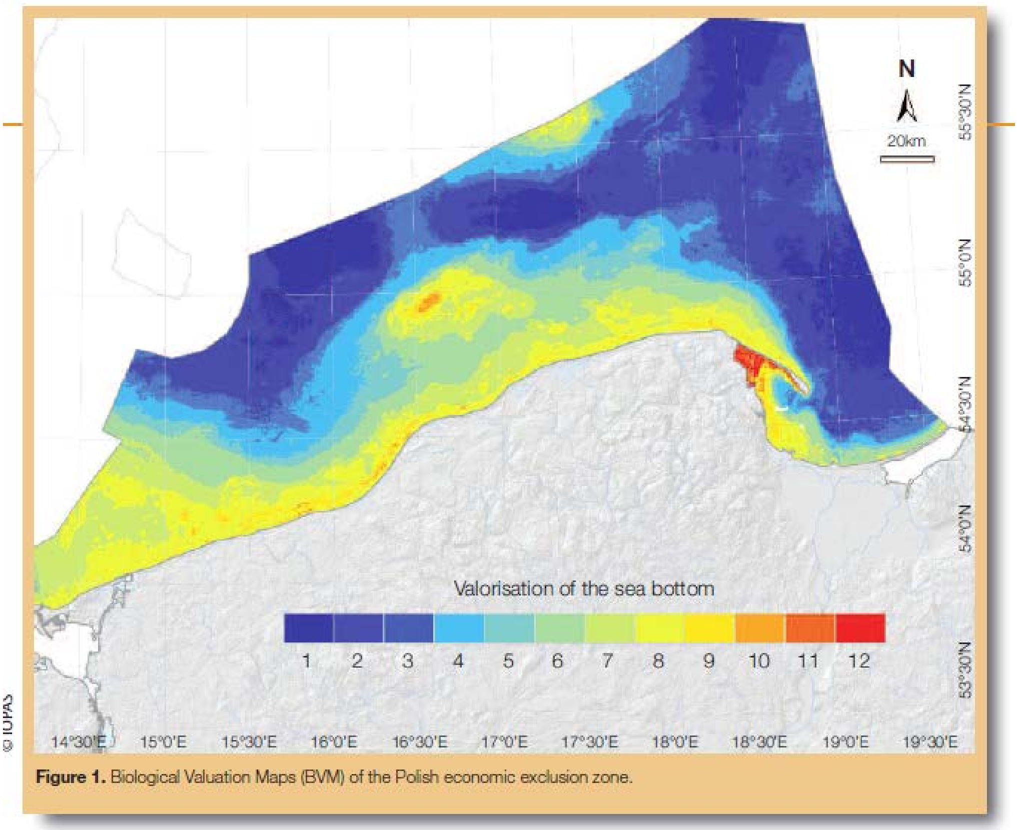

Figure 1. Biological Valuation Maps (BVM) of the Polish economic exclusion zone.

Figure 4.4. Biological valuation map of the Polish EEZ (from

http://www.marbef.org/documents/glossybook/MarBEFbooklet.pdf.)

Case study 2: Biodiversity in the Dutch EEZ.

An important contribution to the achievement of GES is formed by spatial protection measures, as described in Article 13.4 of the MSFD. To provide basic information for such measures, Bos *et al.* (2011) identified biodiversity hotspots in the Dutch EEZ, by the spatial application of criteria for GES Descriptor 1 'Biological diversity is maintained'. They did not focus on indicator species, but on the species assemblages, for which data were available on the scale of the Dutch part of the North Sea. Such data were available for benthos, fish, seabirds and marine mammals and for habitats. On the basis of the criteria, a number of biodiversity metrics were defined for benthos, fish, seabirds, marine mammals and habitats, for which maps were made (Figures 4.5–4.6). For example, for macrobenthos, the metrics included species richness and species evenness, but also the number of species with a maximum age > 10y, the number of species with a body size (> 1 g AFDW) and the frequency of occurrence (called rarity). Values for biodiversity metrics were rescaled on a scale of 1 to 5 and mapped per biodiversity metric per species group. The maps of different metrics were combined, by addition, per species group, using a standardised 5x5 km grid. The combined map was obtained by rescaling the summed values on a scale of 1–5.

The effect of this method was that the more information was combined, the less insight was obtained. The final map was therefore presented as a summary of the hotspots on the separate maps. Difficulties in this approach are the different spatial and temporal scales for different data series, different numbers of metrics per species group, and the scaling of the metrics themselves.

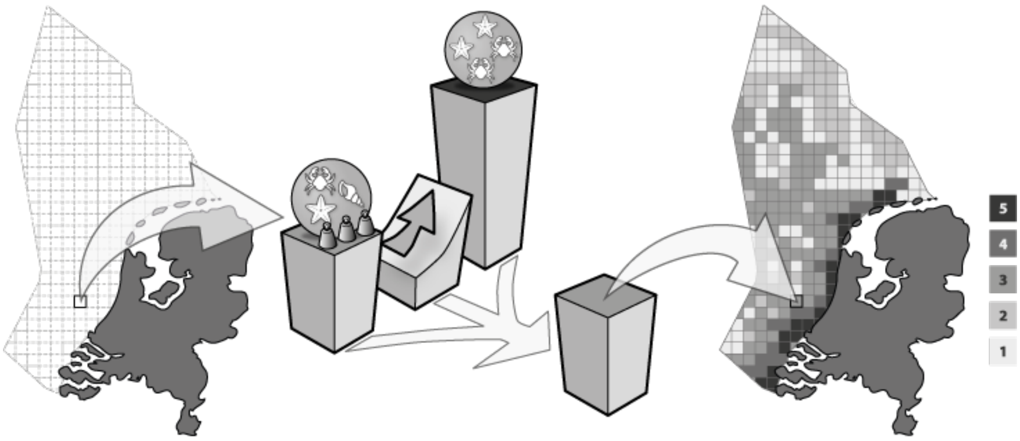

Figure 4.5. Example of the aggregation of 3 biodiversity metrics (biomass, trend, evenness) into 1 metric, by summing rescaled metrics (From Bos *et al.*, 2011).

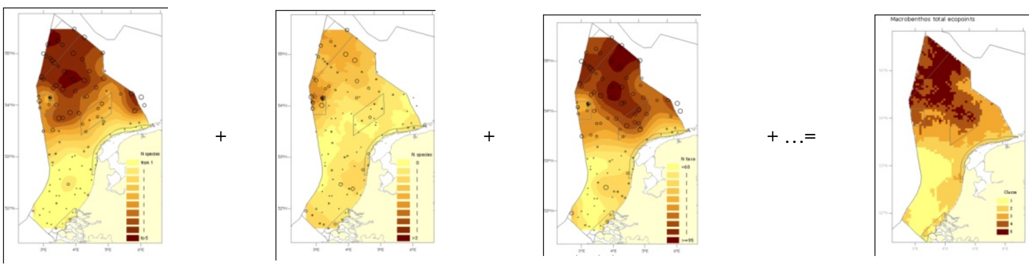

Figure 4.6. Example of summing maps for biodiversity indicators for macrobenthos (From Bos *et al.*, 2011).

4.2.3 Spatial approaches in HELCOM and OSPAR

The Helsinki Commission (HELCOM) and the Oslo Paris Convention (OSPAR) are regionally coordinating the integrated ecosystem assessments between member states in the Baltic and North Sea. Both conventions are the coordinating bodies for the regional implementation of the MSFD across member states. They apply a holistic approach to monitor and assess biodiversity by developing indicators across several taxonomic groups. However, while HELCOM has started to implement an assessment with spatial resolution, OSPAR is just at the start of an integration protocol without spatial resolution.

HELCOM

In 2009 the Helsinki Commission (HELCOM) performed an integrated biodiversity assessment in for the entire Baltic Sea using an indicator-based assessment tool named BEAT (HELCOM 2009 & 2010, see also Section 2). For this assessment, at 73 locations throughout the Baltic the diversity status of marine landscapes, communities and species was assessed. The state of the single case studies was classified from 'high' to 'bad' on a 5-level scale, and the status values have been graphically interpolated and smoothed. It is not fully described how the spatial integration was performed, but there is no evidence that spatial statistics were applied. The HELCOM report on holistic assessment (HELCOM 2010) explicitly states that the BEAT assessment is under construction and should be regarded as preliminary.

OSPAR

The 2010 Quality Status Report by OSPAR demonstrates a pilot approach of integrated assessment by region (OSPAR 2010). The status of fish, birds and mammals was assessed using 22 pressure indicators which were subsequently integrated over a four-level scale. However, currently OSPAR does not address spatial aspects of biodiversity other than mapping the occurrence of vulnerable and protected habitats.

4.3 Caveats of sampling spatial diversity

The basis for assessing biodiversity will be ongoing monitoring and survey programs that will be adapted to the requirements of the MSFD. Most of this will be dealt with in Section 5, but some spatial issues shall be outlined below:

- How can existing surveys be extended to representative habitats which were not sampled previously? For example, sampling reefs and rocky habitats will likely need different gears to those used in existing (soft bottom) trawl surveys? Can data be corrected for habitat- and species-specific catchability?
- When extending the survey to sample new habitat types, existing stations should be retained and not dropped from the sampling scheme. In other words, existing surveys should be extended additively instead of redirecting survey effort to the new areas of interest. Otherwise the time series may become compromised and targets derived from previous time series will be irrelevant to the new data. However, this will have resource implications.
- The raising of data collected on different spatial scales and units should be founded on sound statistical considerations. Extrapolating results obtained from the analysis of samples collected at small spatial scales to larger areas will over-estimate the presence/absence information of rare species and increase the estimated variance. Thus it is more appropriate to scale to an area of comparable spatial scale.
- Diversity 'hotspots' (in terms of species richness) should not be the only areas of interest in marine spatial planning. The focus on 'hotspot' areas risks losing the sight on less diverse yet ecologically important and/or vulnerable habitats (e.g. coral reef *vs.* sandbank).

4.4 Implications for the MSFD

The MSFD addresses spatial diversity on the species, habitat and ecosystem scale (Cochrane *et al.* 2010). Member States are required to report the distribution range, distribution pattern, population condition and habitat range of relevant species. The MSFD also demands the coordination of conservation efforts with other directives (e.g. Habitat Directive, Bird Directive), which implement spatial measures to protect species and habitats. Furthermore, the management of human pressures will be more and more managed on various spatial scales with marine spatial planning increasingly used in many European countries. These plans include the zoning of:

- Shipping and transportation
- Renewable energy
- Marine protected areas (Natura2000 sites)
- Cable and pipeline routes
- Extraction of abiotic resources (oil, gas, gravel, sand)
- Fishing
- Mariculture

Because many of these spatially managed units are defined as pressures in the MSFD, marine spatial planning is tightly linked to the MSFD. Spatial measures to protect diversity, ecosystem function and ensure sustainable use will increase within the current decade and will affect the assessment of the environmental status within the MSFD. Consequently, spatial approaches to map and analyse MSFD indicators will help to integrate pressure and state indicators. Such issues are being addressed by ICES, for example by the Working Group for Marine Planning and Coastal Zone Management (WGMPCZM) and the recent Workshop on the Science for Area-based Management: Coastal and Marine Spatial Planning in Practice, WKCMSP).

4.5 Summary

This chapter presents several studies/methods that have mapped and assessed biodiversity on national/regional spatial scales. These studies were intended to identify biodiversity hotspots, influencing site selection of potential marine protected areas and displaying patterns of biodiversity across regions and subregions. Spatial analysis and modelling of biodiversity are crudely implemented and need further scientific attention. The development of monitoring programs within the MSFD will potentially provide much needed time series on adequate spatial scales in the future.

4.6 References

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5 The implications of survey design for estimating ‘biodiversity’ metrics

5.1 Introduction

The previous meeting of WGBIODIV (ICES, 2010a) included much debate on the implications of survey design on the subsequent derivation of any metric of ‘biodiversity’. Members of WGBIODIV then provided examples of this in a presentation to the Marine Biodiversity theme session at the 2010 ICES Annual Science Conference (Borges *et al.*, 2010).

In essence, monitoring of marine organisms uses particular field sampling techniques and programmes that are often designed to optimise the sampling of a particular species or range of species. All aspects of survey design and the subsequent collection of field data can, therefore, affect derived metrics that may be used to inform on patterns of biodiversity.

Given that such issues are fundamental if the ICES community is to advise on the state of marine biodiversity, WGBIODIV proposed the following ToR "To examine the implications of survey design for estimating 'biodiversity metrics' ".

Given the expertise of the members of the Working Group, the emphasis is on species diversity.

5.2 Overview of the aspects of survey design that may affect biodiversity metrics

5.2.1 Gear selection

All sampling gears have their advantages and disadvantages, with some species (or size categories of species) sampled more effectively than others, and some species may never be captured by the gear in question. Gear selection for a survey can be influenced by several factors, including effectiveness of sampling the target taxa/habitat, reliability, cost, durability, availability and consistency (e.g. in the case of internationally-coordinated surveys or in order to maintain a time series).

The multi-taxa data collected from gears such as trawls, dredges and grabs are often used to inform on the diversity of fish and invertebrates. In reality, not all 'fish' or 'invertebrate' taxa are sampled effectively, and the data do not necessarily inform on what species are present at the 'community' level. Rather, such data inform on the broad assemblage of organisms that are sampled at that time and by the gear used. Besides one of the original aims of these surveys were the monitoring of commercial fish stocks, and this aim has affected many of the decisions on the design of the surveys.

The gear used in bottom trawl surveys is similar to those used by the commercial vessels, but modified to sample more adequately the target species, especially recruits, and reduced in size to obtain "manageable" samples.

5.2.2 Timing of sampling

Many of the large-scale, offshore surveys are undertaken annually, and in most instances the timing of the survey is kept consistent from year to year. Usually the decision on the season of sampling was driven by the recruitment or spawning of target species. It should be recognised, however, that some species may have important seasonal differences in their abundance and distribution, which may be due to migratory patterns or recruitment events.

5.2.3 Site selection

There is considerable debate with regards to optimal sampling grids. The survey grids sampled in most offshore surveys are usually based on either a fixed station grid or on a random stratified design, although often the latter is based on the random selection of stations known to be suitable for sampling.

In addition to the advantages/disadvantages of the survey grid selection, other factors influence survey design, which may subsequently limit the utility of the data for species diversity studies. In terms of regional biodiversity assessments, it should be recognised that some grounds may not be sampled, for example:

- Sites in shallow water or in proximity to other shipping hazards are, for obvious safety reasons, usually excluded from the survey grid. This has implications for the effectiveness of sampling coastal and estuarine fish.

- The original rationale for the survey may dictate that the density of sampling stations is low on grounds that are viewed of lesser importance.
- The gear may not be suitable for use on certain grounds (e.g. depending on water depth, substrate type, tidal stress).

5.2.4 Density of sampling stations

The density of stations is rarely considered in offshore surveys, and when it is considered is for the assessment of the target species, not to consider biodiversity issues. For example, the North Sea IBTS will attempt to ensure that 2–3 stations are fished in each ICES rectangle, and other surveys will also be stratified by bathymetry.

5.2.5 Sample replication

Benthic studies typically use (3–5) (pseudo) replicate samples from with which to derive data on the invertebrates at a site. In contrast, fishing surveys generally conduct a single tow at the sampling site.

In terms of species richness (an important element of 'biodiversity monitoring'), sample replication is a major issue, even for towed gears. It is well established that the number of species observed increases with increasing sampling effort, and so it could be asked how reliable any single tow is for the purposes of monitoring a diversity metric for fish or epifauna. It has been shown that at least 20 IBTS trawls are necessary to estimate alpha-diversity at the scale of an ICES rectangle (Greenstreet and Piet, 2008).

Ellis *et al.* (2007) examined the number of species captured by 2m beam trawl in selected habitats, including mud (Celtic Deep and North-west Irish Sea), sand (inner and outer Carmarthen Bay and the outer Bristol Channel), sand banks (Swarte Bank and Broken Bank in the southern North Sea), gravel (off the Suffolk coast) and shell gravel (in the western English Channel). Shell-gravel habitats in the western English Channel had the highest species richness, and the asymptote was not reached after 13 tows at the site (Figure 5.1). Even sand bank habitats in the southern North Sea, which had the lowest observed species richness, would need extensive sampling to monitor species richness effectively.

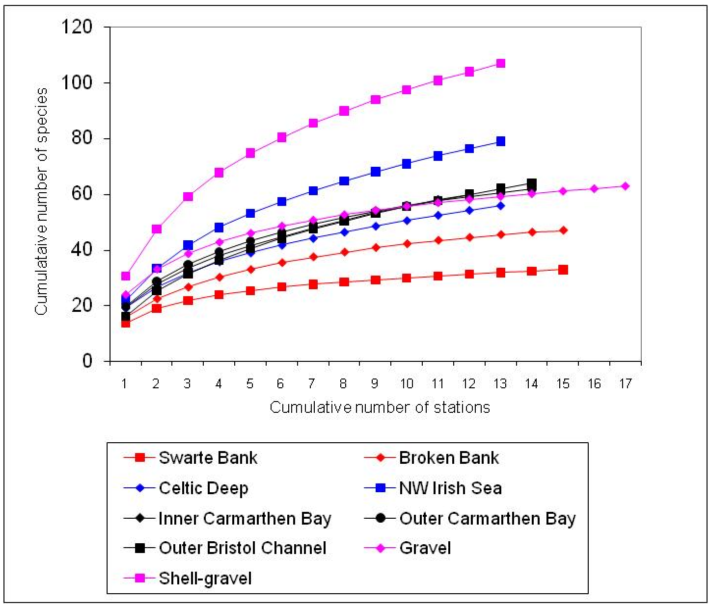

Figure 5.1. Cumulative number of species taken whilst undertaking 2 m beam trawling in selected habitats around the British Isles (Adapted from Ellis *et al.* 2007).

5.2.6 Catch processing

Trawl surveys generally attempt to either (a) sort the entire catch to species (or appropriate higher taxon), or (b) process the entire catch for 'rare' species and sub-sample the remaining 'mix' to get appropriate catch components for the main species. The latter is used when there is a large and/or complex catch.

Very rarely, some trawl surveys will only sub-sample a large catch, and so this method has the possibility of missing a 'rare' species or, if it is found, applying an unrealistic raising factor in the estimated catch. Such processing, although time efficient, are not suitable for use in community/biodiversity studies.

5.2.7 Taxonomic resolution

Taxonomic expertise is an important 'variable', especially when data sets are combined across laboratories, years and/or regions. For example, some less qualified staff may not fully realise the difference between recording species at genus or family level (e.g. it has been noted that some sea-going staff consider *Ammodytes* spp. and Ammodytidae to be inter-changeable, when in fact they are subtly different).

Given that much identification is done with either 'gestalt' recognition for large/obvious taxa, and dichotomous keys for other taxa, the accuracy of the identification is reliant on the person having an appropriate knowledge and that any identification keys used are sufficiently up-to-date and cover all relevant species for the geographic and bathymetric region being surveyed. Nevertheless, quality assurance and quality control measures should be in place, including appropriate taxonomic training courses and ring-tests between laboratories.

5.2.8 Data filtering and standardisation

Given many of the limitations discussed above, especially with regards gear selection and taxonomic resolution, there may be some justification for filtering data (e.g. to combine problematic species at a higher taxonomic level, or to exclude species that are not considered to be sampled effectively).

However, this is highly reliant on 'expert judgement' and rarely justified as to why which species were excluded. When such filtering is used it is advisable to document the filtering done, and the rationale behind the filtering, especially when considering comparisons with other areas and in historical studies.

5.3 IBTS surveys

IBTS surveys, being a survey with a time series that started in 1961, and that adopted a standardized methodology from the early 1970s (ICES 2010b and 2010c IBTS Manuals, although most standardized since 1983 (for Quarter 1), and later for Quarter 3), the IBTS has been considered as an important source of data for monitoring fish stocks and, more recently, biodiversity. Nevertheless many of the caveats and problems mentioned above have to be carefully considered when using these data for biodiversity studies:

- The original aim of the survey was to provide information useful for fish stock monitoring and management (i.e. in support of the Common Fisheries Policy). The survey was not designed to assess species diversity, this poses caveats for the use of the IBTS survey for diversity studies, and these have to be acknowledged and considered when conclusions are drawn.

  Nevertheless, IBTS surveys have a relatively broad time series, which can give the option to describe potential baselines and reference values in terms of potential species diversity indices (e.g. using the large fish indicator as a surrogate to monitor biodiversity for demersal fish).
- Taxonomic expertise can be variable. It is important to include in the survey protocols and manuals the importance of asking an appropriate taxonomic expert when rare organisms are found, and to have careful procedures to document and keep records of such species to confirm identification. It is also important for the management of the surveys to ensure that taxonomic expertise is on board. It is also very important that the effect of such taxonomic expertise, which may vary over time and between national surveys, is considered in biodiversity studies (see Daan 2001; ICES 2007a). Examples of this are also shown in Section 3.
- The issues of tow duration vs. numbers of tows was covered by WGBIODIV last year (ICES 2010a), and also expanded in Borges et *al.* (2010).
- A common practice with the use of IBTS abundance indices, especially those used for assessment of commercial stocks purposes (e.g. by the various ICES stock assessment Working Groups), is to provide indices in abundances per hour trawled, since the fishery effort is usually reported in hours trawling or days at sea. This practice is not deemed adequate for diversity studies since it will overestimate the numbers of rare species/individuals (given that most surveys now fish for 30 minutes, and so numbers are doubled to ind.h^{-1}). For diversity studies, it may be more appropriate to conduct analyses of the abundance per swept area, and using area units that are close or equal to the original sampling area unit, and not raising to much higher spatial units.
- Within the MSFD there is a descriptor to address specifically the status of commercial stocks (Descriptor 3: "Commercial fish and shellfish exhibiting a population age and size distribution that is indicative of a healthy stock"). In this sense it is important that when considering indices of evenness and richness, commercial species should still be retained, in addition

to non-target fish, including species that may be considered as vagrants or at the limits of the distribution range.

- On the other hand, when considering other indicators based on groups of species (i.e. the proportion of large fish) where the indicator estimation procedure has been considered for a particular group and area (e.g. for demersal fish in the North Sea), the targets will have to be redefined for other areas considered, and different species may have to be selected/filtered, or analysed separately, with well documented protocols (e.g. it may be that pelagic species, or fish species that are narrow-bodied and elongated need to be excluded from further analysis, but the rationale for their exclusion documented).
- Fish surveys aim to monitor the strength of annual recruitments, and sometimes there are concerns from the fishing industry that the surveys are not sampling areas "where the (adult) fish are". This can lead to pressure to sample different stations or to increase sampling effort in some areas. This will affect diversity studies and the time series. If a change in survey grid is required, this should be done by adding the extra stations (for better understanding the species of interest) but not considering these stations in diversity studies since it may bias the continuity of the time series. Re-directing survey effort away from sites where commercial species are less abundant to other areas has implications for diversity monitoring that should be considered.

5.4 Case study of the Spanish Porcupine Bank Survey

The Porcupine Bank survey is carried out annually, covering an area from 12–15°W and from 51–54°N, and a depth range of 180–800 m. The cruises are carried each September on the RV "*Vizconde de Eza*", a stern trawler of 53 m and 1800 kW, and the survey is coordinated by, and following the protocols of, the ICES International Bottom Trawl Survey Working Group (ICES, 2010c). The stratification used in the surveys is based in the distribution of faunal assemblages and the main commercial species in the Porcupine Bank, combined with bathymetry information provided by the National Geological Survey of Ireland.

The main survey aims are to obtain stratified abundance indices and annual recruitment strength for the commercial species in the area (hake *Merluccius merluccius*, anglerfish *Lophius* spp., megrims *Lepidorhombus* spp. and *Nephrops norvegicus*), to estimate the strength of their recruitments, to collect biological information, and to study and describe the distribution patterns of the faunal assemblages in the area.

80 trawl stations are sampled every year, and following the IBTS protocols the entire fish catch is sorted to species, wherever possible. Representative samples of all fish species are measured to obtain the length distributions. Thus the information collected in the survey is adequate for developing a "large fish indicator" (LFI), as proposed by Piet *et al.* (2007) as an indicator reflecting the size structure and life history composition of the fish assemblage. For this indicator "large fish" has been defined as >40 cm (ICES 2007b), and an EcoQO reference limit was defined by OSPAR to be 0.3 or greater, this value was defined based on the value in the early 1980s in the North Sea, when fish stocks were not thought to be suffering from widespread over-fishing (Greenstreet *et al.* 2011). However it is interesting to note that these values were determined on the basis of survey information from the North Sea, thus the LFI reference value and the size to define 'large fish' needs to be explored for different areas and surveys covering different ecosystems. The Porcupine Bank survey covers deep

grounds (180–800 m) and many of the fish species captured include deep-water species with relatively large sizes. Therefore it offers the opportunity of exploring the application of the indicator to different areas and ecosystems.

Calculation of the "large fish indicator" (LFI) is based upon fishery independent trawl survey data that reports CPUE of species by length applying the formula:

W≥ 40cm / Wtotal

As individual weight was not recorded for most fish species examined, length-weight (L-W) relationships were used to estimate individual fish weights. The L-W formula used was:

Weight = a.Length^b

Values of *a* and *b* were taken from survey data where possible, and were obtained from FishBase (www.fishbase.org) for other species.

Macrourid species were excluded *a priori* since their shape length/weight is not comparable with the rest of the species considered. Some other infrequent species (ca. 40) were also excluded for this case study, as no length-weight relationship data were available. However, it should be noted that these species accounted for <0.1% of the mean stratified weight in the survey.

Additionally, the LFI was also calculated excluding pelagic species to reduce the possible effect of highly variable catch rates of certain species, following the rationale used by Piet *et al.* (2007) that "*The indicator can be calculated for the entire assemblage that is caught by that particular gear or a subset based on morphology, behaviour or habitat preferences (e.g. bottom-dwelling species only)*". The species excluded as pelagic (or benthopelagic) were Ray's bream *Brama brama*, boarfish *Capros aper*, snipefish *Macrorhamphosus scolopax*, horse mackerel *Trachurus trachurus*, mackerel *Scomber scomber* and sardine *Sardina pilchardus*.

Figure 5.2 shows the trends in the LFI indicator along the 10 years of the current time series, using different lengths to define large fish (40, 30 and 20 cm) and also including or excluding pelagic species. According to these results, the LFI has increased during the ten years covered by the survey time series. This result coincides with those found by Greenstreet *et al.* (2011). Nevertheless the EcoQO (0.3) is not met when using 40 cm to define large fish, but it increases from values close to 0.1 in the first years to a value of 0.27 in 2010. The same overall image is found when using 30 cm to define large fish though the indicator meets the EcoQO at the beginning of the series (2001) then decreases below 0.3 between 2002 and 2005, reaching again the EcoQO in 2006 and going beyond 0.4 for 2008–2010. On the other hand when using 20 cm to define large fish the LFI clearly enters the length range where the noise created by the recruitments decreases the sensitivity of the indicator.

Figure 5.3 shows the cumulative length distributions sampled for the total time series, showing the percentile reached with each length used to define large fish (only shown here with pelagic species included), >40 cm: 1.1%, >30 cm: 6.5% and >20 cm: 59%. Therefore, in the case of the Porcupine Bank survey, a 40 cm length seems a suitable limit to define large fish in this area/survey.

Regarding the exclusion of pelagic species, no differences were noticeable with the exclusion of pelagic species, most probably because the study area is away from the Irish coast and shallower grounds, and the Irish shelf is separated from the Porcupine Bank by the Slyne Ridge, a trench that splits the Porcupine Bank from the Irish shelf with depths of about 350 m, which may hinder the entrance in the area of schooling species from the Irish shelf.

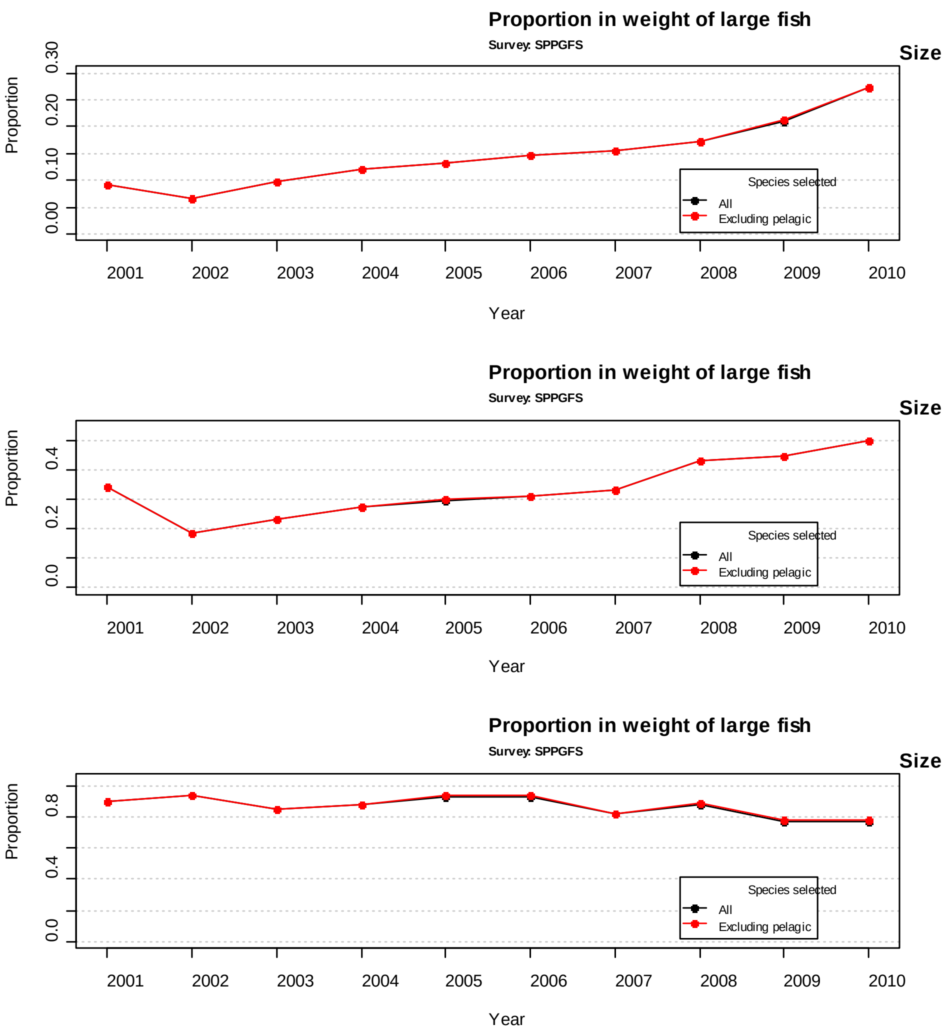

Figure 5.2. LFI variation using three different length thresholds (top to bottom: 40 cm, 30 cm, 20 cm) to define large fish, and including pelagic fish (black line) and excluding pelagic fish (red line). Data from the Spanish Porcupine Bank survey (2001–2010).

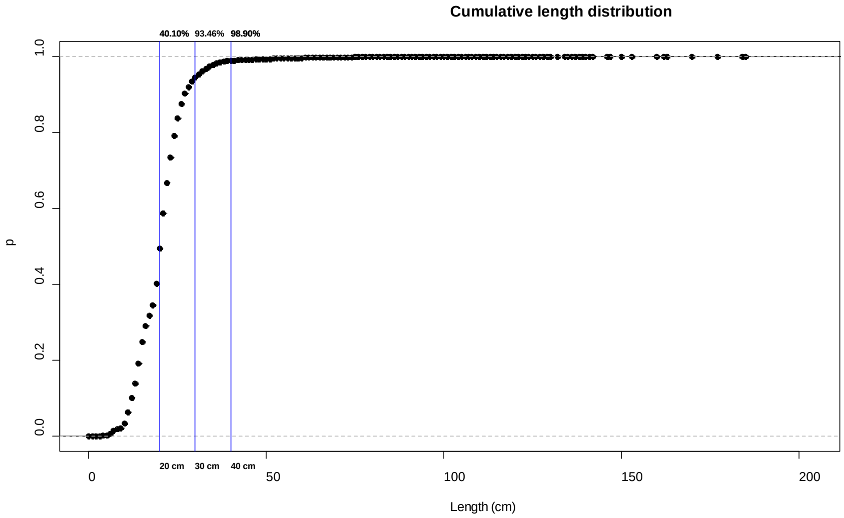

Figure 5.3. Cumulative length distribution using all the time series data.

The decisions about the exclusion of species without L-W parameters, pelagic species or macrourids are examples of the near-arbitrary nature of species selection, which may limit its relevance to diversity studies. When a biodiversity-related index is not addressing species richness, the importance of these decisions can be assessed through the proportion of the total fish biomass excluded from these decisions. In the Porcupine Bank example, the only exclusion that could be considered important was for macrourids, as this group has increased in relative biomass from 1% of the fish biomass in 2001 to more than 10% in the last four years (2007–2010).

Of the other issues mentioned above, it must be recognised that the original survey aims were not for diversity sampling. In the case of the Porcupine Bank survey, monitoring the abundance of commercial fish species is the primary aim, and the increase in the relative abundance of hake, especially the larger ones found in the first year and in 2009/2010, is an important driver for the overall result of the LFI, since in 2010 hake larger than 40 cm made up the 34% of the total biomass of fish larger than 40 cm, and three species (hake, anglerfish *Lophius budegassa* and greater forkbeard *Phycis blennoides*) comprised about 50% of the total biomass larger than 40 cm in all the years studied (Figure 5.4).

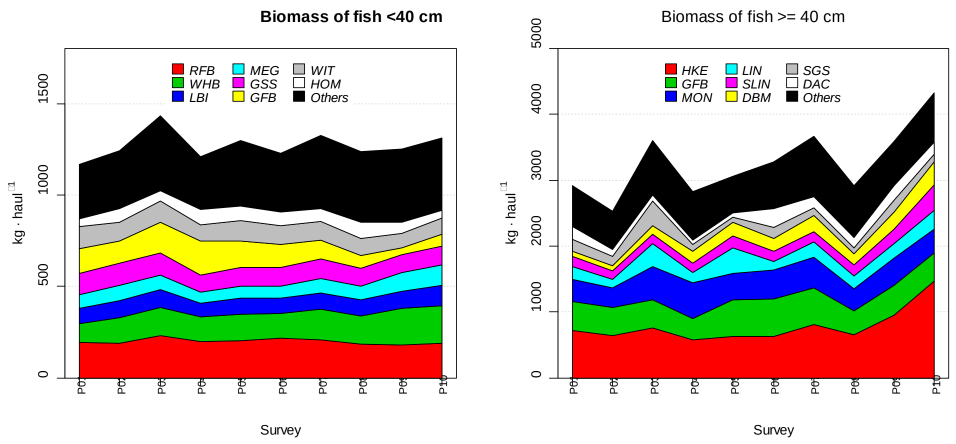

Figure 5.4. Main species contribution to biomass of fish smaller than 40 cm (left) and larger than 40 cm (right). For species see Table 5.1.

Table 5.1. Species codes used in figure 5.4.

Code	Common name	Scientific name
RFB	Blackbelly rosefish	*Helicolenus dactylopterus*
WHB	Blue whiting	*Micromesistius poutassou*
LBI	Fourspot megrim	*Lepidorhombus boscii*
MEG	Megrim	*L. whiffiagonis*
GSS	Greater smelt	*Argentina silus*
GFB	Greater forkbeard	*Phycis blennoides*
WIT	Witch	*Glyptocephalus cynoglossus*
HOM	Horse mackerel	*Trachurus trachurus*
HKE	Hake	*Merluccius merluccius*
MON	Monkfish	*Lophius piscatorius*
LIN	Ling	*Molva molva*
SLIN	Spanish ling	*Molva Macrophthalma*
DBM	Blackmouth catfish	*Galeus melastomus*
SGS	Sixgill bluntnose	*Hexanchus griseus*
DAC	Birdbeak dogfish	*Deania calcea*

5.5 Ecosystem surveys

There are questions on how to cover other biodiversity components, as well as those species and habitats of conservation importance that are not sampled during existing surveys. Existing surveys should be maintained as standardized surveys, so as to maintain consistency in current biodiversity information, although additional new aims/sampling could be introduced to these surveys (depending on resources). Any changes in survey protocols should be documented in survey manuals so as to ensure future scientist have the relevant information with which to interpret biodiversity data.

Certain components of the ecosystem could be covered within various on-going surveys, but the appropriateness of the survey methodology to the new components needs to be considered to optimize the use of resources. For example, would seabird and marine mammal monitoring be better undertaken in acoustic surveys (where transects are followed) rather than on groundfish surveys, where the potentially continuous process of fishing and catch processing may attract seabirds. Epibenthic sampling with 2 m beam trawl was successfully integrated in the North Sea IBTS for a number of years (Callaway *et al.*, 2002 and references cited therein), although the deeper waters of the continental shelf of the ICES area means that such sampling can be more time consuming (in terms of deploying and retrieving the gear), which then may result in more limited sampling opportunities during daylight. Nevertheless, there have been several surveys using small beam trawls to examine epibenthic assemblages in the Celtic Sea and Iberian waters (Ellis *et al.*, 2002; Serrano *et al.*, 2006, 2008).

In addition to IBTS surveys, there are beam trawl surveys for fish in the North Sea, English Channel, Bristol Channel, Irish Sea and parts of the Bay of Biscay and parts of the Celtic Seas. These gears can catch large volumes of epibenthic material, and whereas the data from these surveys can inform on spatial patterns in demersal assemblage structure (e.g. Kaiser *et al.*, 1999; Ellis *et al.*, 2000), the large degree of subsampling of the epibenthic bycatch that may be involved (unless there was to be a large increase in staff resources) hampers accurate collection of biodiversity information (in terms of species richness etc.).

It should also be noted that many of these surveys operate on 'trawlable' grounds. Many North Sea IBTS vessels operate on finer grounds (so as to minimise gear damage), with some IBTS nations using trawls with rockhopper ground gears to allow coarser grounds to also be sampled. Similarly, the use of a chain mat on some beam trawls allows these surveys to operate on coarse grounds. Nevertheless, there are some stony grounds that are not realistically suitable for trawl survey, and if such habitats are to be surveyed this would involve very different sampling methodologies, including non-destructive sampling techniques if associated with biogenic habitats (or if in areas designated as some form of MPA).

IBTS and beam trawl surveys process the catches at sea, with land-based work including otolith processing and identification/confirmation of unusual fish. The expansion of work to include epibenthic fauna would involve further samples being brought ashore for accurate identification (catches themselves can be processed and many of the species fully recorded at sea). Hence, there is a requirement for post-survey resource. If trawl surveys are used as platforms for the sampling of other ecosystem components, especially for small-bodied groups (e.g. plankton, parasites, hyperbenthos, infauna, meiofauna, microbes etc.), then although cost-effective field sampling can be undertaken (in terms of ship time), there are major implications for the original survey (e.g. in terms of extra gear deployments) and for the extra disciplines (in terms of the resource required to process the samples in the laboratory).

The move towards ecosystem surveys is being addressed by ICES, through the Working Group on Integrating Surveys for the Ecosystem Approach (WGISUR), although the report from the 2011 meeting of this expert group was not available when WGBIODIV met.

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6 The capacity of the ICES science community to address key issues of 'biodiversity science'

6.1 Introduction

WGBIODIV were requested to "liaise with the ICES Strategic Initiative on Biodiversity (SIBAS) to identify the potential capacity of the ICES science community to address key biodiversity science issues and provide biodiversity advice", with emphasis on the following issues:

(i) Biodiversity and ecosystem services: the role of biodiversity in supporting ecosystem services and the social and economic consequences of human impacts on biodiversity.

(ii) Diversity and ecological processes: The extent to which the diversity of a community influences (a) 'stability', (b) productivity, (c) resistance to invasion or disease, and (d) ability to recover from natural and human impacts, and interactions between these factors. The changes in production among systems that differ in biodiversity. The role of biological invasions in altering system production and energy flow.

(iii) State of biodiversity: patterns and trends in biodiversity and the structuring roles of evolution, ecology and environment.

(iv) Functional significance of biodiversity: the functional significance of genetic, species, population and ecosystem diversity. Redundancy and the extent to which species in a functional group are interchangeable. Comparisons of system function and biodiversity.

(v) Measuring biodiversity: measurements of genetic, species, and ecosystem biodiversity and the errors associated with these measurements. The effects of errors on understanding of ecosystem structure and function.

(vi) Biodiversity futures: projecting future changes in biodiversity in response to projected human and environmental drivers.

6.2 Biodiversity and ecosystem services

6.2.1 The role of biodiversity in supporting ecosystem services

How biodiversity supports 'ecosystem services' is poorly understood for many marine environments. There are several forms of ecosystem services, including:

a) Supporting, including the provision of ecologically important biotopes (e.g. refuges, habitat formation that may minimise impacts of natural events such as flooding), primary production, nutrient cycling and water quality;

b) Provisioning of food, materials (e.g. pharmaceutical and biochemical products, aggregates), genetic resources and energy (including space for energy developments);

c) Regulating, including the oxygenation of sediments, decomposition of waste materials, biological regulation of populations, climate and disturbance (e.g. erosion) regulation, as well as gas exchange;

d) Cultural (including recreational) benefits.

Links between ecosystem services, their economic value and relation to ecosystem function were summarised by Costanza *et al.* (1987).

For some of these services, biodiversity plays an integral part of the provision. Habitat diversity provides crucial and productive habitats, but also allows for usage including energy provision or for material extraction. The discovery of novel biochemical substances is strongly linked to diversity, based on niche partitioning and adaptation to pathogens, as is the provision of genetic resources. Nutrient cycling supported by bioturbation and pollutant decomposition, for example, are provided by a variety of differently adapted organisms. The latter strongly relies on genetic diversity to enable degradation of novel compounds based on existing metabolic pathways.

Keystone species are those species whose effect on an ecosystem is disproportionally large relative to their abundance (Power *et al.* 1996) and they play an important role in maintaining the structure of a habitat and/or food web. Foundation/engineering species play major roles in shaping communities by creating and enhancing habitats in ways that benefit other species. Examples are primary producers such as kelp and eelgrasses creating forests and seagrass meadows (Miller and Spoolman, 2009).

Keystone and foundation species play similar roles. The major difference is that foundation species help to create habitats and ecosystems whereas keystone species play an active role in maintaining the ecosystem and keeping it functioning in a way that serves the other species living there. Examples of species considered as keystone species are sea otters in the kelp forest (Estes *et al.* 1978) and starfish in mussel beds (Paine 1966, 1969). Sea otters predate on sea urchins, thus preventing them from overgrazing the kelp that is the foundation species in the kelp forest that allows other species to thrive. Without the kelp there will not be an optimal habitat for many other species, and it thereby plays a critical role in maintaining the structure of the environment. The same is true for starfish, without them the mussels may produce extensive beds, and such 'monoculture' may exclude some other species.

The identification of foundation and keystone species is crucial for ecosystem management to maintain the stability of ecological communities.

The transfer of the theoretical concept of 'ecological redundancy' into practical assessments is problematic. A species might appear functionally redundant (i.e. it has the same function as another species), but we have a very limited knowledge on functional variability under different environmental conditions, which prevents a full assessment of redundancy (Wellnitz & Poff 2001). These authors also stressed that functional redundancy is likely to be context-dependent, which limits the transferability of this theoretical concept to practical management purposes. For example, although two species may undertake the same role in one function, they may have very different roles in other functions, e.g. trophic position and contribution to nutrient cycling (e.g. Walker, 1992).

Although redundant species are sometimes referred to as at least partly substitutable (Loreau *et al.* 2002), it is thought that 'redundancy' in biodiversity contributes to the resilience of ecosystems (Naeem 1998). Having apparent 'redundancy' simply ensures that ecological functions are carried out under a range of environmental conditions (Hooper *et al.* 2005; Section 3.5 of ICES, 2010a). Despite the challenge, testing this theoretical concept in experimental studies on microorganisms have been successful in showing redundancy (Langenheder *et al.* 2006). In contrast, a strong link between functional changes and variation in species diversity indicates low levels of redundancy in other groups, as shown for reef-fish assemblages (Micheli & Halpern 2005; Halpern & Floeter 2008).

Currently, ICES has limited capacity to examine the redundancy concept under field conditions, and to quantify how 'biodiversity' contributes to some ecosystem services.

6.2.2 The social and economic consequences of human impacts on biodiversity

ICES has not traditionally undertaken work in socio-economics, although there is greater interest in this field, especially in terms of fisheries economics. There are also potential aspects of biodiversity that may be linked with other aspects of socio-economics.

Economic analyses of human activities are fairly novel approaches to assess impacts on the marine ecosystem. Here we provide a few examples on which there are clear links between impacts on biodiversity and economic and social effects.

Reduced habitat diversity may affect the health, quality and size of fish populations. Damage to ecologically-important habitats (e.g. by fishing activities, contamination, dredging) can affect some of the fish living in the habitat, for example in response to changes in food resources and habitat quality (e.g. access to refuges from predators). There have been some recent developments in fisheries management aiming to take the economic consequences of habitat degradation into account (e.g. Turner 1999; Thrush & Dayton 2002; Bradshaw *et al.* 2003; Carbines *et al.* 2004; Grafton *et al.* 2006, 2009; Scharf *et al.* 2006; Armstrong & Falk-Petersen 2008).

It should be noted that some human impacts, such as habitat modification (e.g. addition of hard substrata into the marine environment) can result in major changes to community structure although without a decrease in overall species richness (in terms of the number of species present).

Recreational activities (e.g. angling, diving) may also be affected by biodiversity loss (e.g. Carpenter *et al.* 2008).

Economic impacts may be caused by the 'non-discovery' of potential pharmaceuticals due to biodiversity loss. This is difficult to assess, but it has been suggested that a decreased biodiversity, especially for certain groups of organisms such as sponges, will contribute to a decreased likelihood of pharmaceutical discoveries.

There is some dispute as to what extent harmful algae blooms can be related to human impacts; although researchers worldwide are involved in respective assessments. For an overview of this see Glibert (2007), Anderson *et al.* (2008) and reports from the ICES Working Group on Harmful Algal Bloom Dynamics (WGHABD) and the Study Group to Review Ecological Quality Objectives for Eutrophication (SGEUT).

Invasive species can have various impacts on ecosystems with the largest effects being caused by habitat modifying organisms (Crooks 2002). Some invasive species decrease habitat heterogeneity and thus decrease species abundance and richness. Shipping and aquaculture activities are seen as major sources for the introduction of invasive species. Taking into account species introduced via these routes Molnar *et al.* (2008) developed an ecological impact score system to distinguish various effects caused by certain invaders. Important examples of invasive species are the Chinese mitten crab *Eriocheir sinensis* in Northern Europe, North America and the Mediterranean and the shipworm *Teredo navalis*. ICES is well placed to comment on the impacts of such species, through the work of the Working Group on Ballast and Other Ship Vectors (WGBOSV) and Working Group on Introductions and Transfers of Marine Organisms (WGITMO).

6.3 Diversity and ecological processes

6.3.1 The effects of diversity on the stability, productivity, resistance and recoverability of communities and ecosystems

In ecosystem science, high levels of redundancy and thus high species diversity are considered to contribute to ecosystem resilience which can be used as a proxy for stability and resistance (see Section 3.5 of ICES 2010a, and above). Whilst resistance has been defined as the "*tendency to withstand being perturbed from the equilibrium*" (Connell & Sousa, 1983), ecosystem recoverability is often regarded as "*The ability of a habitat, community or individual (or individual colony) of species to redress damage sustained as a result of an external factor*" (http://www.marlin.ac.uk). Different ecosystems will have varying degrees of recovery potential depending on their abiotic and biotic structures, the impact itself as well as its intensity (Elliot *et al.* 2007).

It is often assumed that systems with high redundancy will have generally high recoverability, however, stressors selectively targeting ecologically important groups within a community on a large scale, may impair redundancy in such a way that recoverability can be largely reduced (Worm *et al.* 2006). For example, benthic habitat recoverability depends largely on physical properties (e.g. muddy sands and reef habitats showing high vulnerability) when subjected to a large scale stressor like fishing (see Foden *et al.* 2010). Also the complexity of food webs may also contribute substantially to recoverability (Dunne *et al.* 2004).

The relationship between productivity and species diversity appears more complex. It has been widely studied in terrestrial and freshwater systems (Fridley 2001, Cardinale *et al.* 2009; He & Zhang 2009 and references therein) giving rise to the assumption that productivity drives diversity and *vice versa*. As environmental conditions have a great influence on the relative contributions of species within a community it is difficult to generalise the relationship between species diversity and ecosystem productivity (Cardinale *et al.* 2000). However, general links could be shown for benthic communities (e.g. Rex 1976; Levin *et al.* 2001; Covich *et al.* 2004; Hiddink *et al.* 2006; Witman *et al.* 2008; Corliss *et al.* 2009; Escaravage *et al.* 2009), taking into account that strong correlations with other factors (e.g. habitat heterogeneity, depth and temperature) exist. The specific shape of the relationship between productivity and species diversity is highly variable including high diversity linked to intermediate productivity, but also linear relationships have been reported. It is thought that this can be related to differences in spatial scales (e.g. Chase & Leibold 2003) and variation in adaptation (Yachi & Loreau 1999). Also inter-specific interactions accelerated by higher species diversity can increase performance of communities e.g. resource consumption (Cardinale *et al.* 2002).

Overall, it is challenging to link ocean productivity and biodiversity on a global scale. Firstly, data on productivity are mainly based on surface primary production (Figure 6.1), and so does not take into account subsurface, coral reef and deep-sea production, with implications on different temporal and spatial scales.

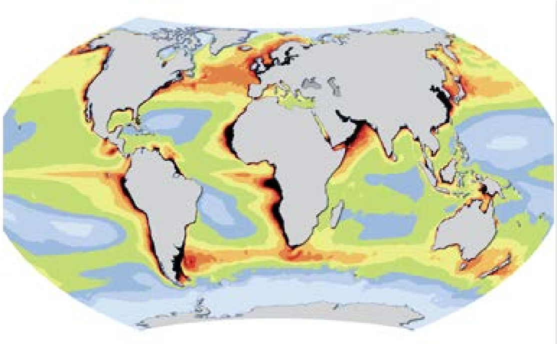

Figure 6.1. Map displaying the most productive sea areas using annual Net Primary Productivity, which are linked with the high biodiversity and biomass (From Nellemann *et al.*, 2008).

Within the ICES area, data on primary production with high temporal resolution exist from various long time series with major focus on coastal areas (see www.wgpme.net for more details), and ICES Expert Groups should be well placed to examine how spatial patterns in the diversity of certain groups is related to productivity and other environmental factors.

6.3.2 The role of biological invasions in altering system production and energy flow

A biological invasion occurs when a species enters and rapidly colonises (through multiplying and spreading) a new area. In many instances, biological invasions have been associated with human activities (e.g. introduction of non-native species for commercial purposes e.g. species for mariculture, accidental means e.g. ballast water or release/escape from captivity).

However, it is also possible for some native species to rapidly increase in abundance and/or distribution in parts of their overall biogeographical range where they would typically be uncommon (e.g. sub-optimal habitats), and such events are also often referred to as biological invasions. This has been documented for species such as blue-mouth redfish (Heessen *et al.* 1996) and snake pipefish (Kirby *et al.* 2006; van Damme & Couperus, 2008).

The importance of such invasions for ecosystem production and energy flow will depend on the position of respective invaders in the food chain and their niche within the ecosystem in question. For example, non-native predators can have evolutionary effects (e.g. Freeman & Byers 2006), whilst the cycling of nutrients can be affected by invasive primary producers (e.g. Larned 2003). In contrast, the introduction of ecosystem engineer species may not necessarily change food web structures (e.g. Brusati & Grosholz 2009).

ICES has strong expertise in this topic, specifically within WGITMO. However, impacts related to food webs and ecosystem energy budgets appear greatly understudied.

6.4 State of biodiversity

6.4.1 Patterns and trends in biodiversity

The ICES community is well placed to describe spatial (e.g. regional, latitudinal, bathymetric) patterns in biodiversity for some taxa, especially in terms of continental shelf habitats.

ICES coordinated surveys also allow temporal changes in 'biodiversity' of some groups (e.g. demersal fish) to be examined. The use of historical information can also be used to infer longer-term changes. The availability of survey data was summarised in the 2010 report of WGBIODIV for microorganisms, meiofauna, benthic infauna, benthic epifauna, benthic habitats, ichthyofauna and cephalopods (See Section 2 of ICES 2010a).

For more detailed information on the status of biodiversity in certain regions, the reader is referred to the work of the ICES Working Group for Regional Ecosystem Description (WGRED) and regional Quality Status Reports (e.g. HELCOM 2010; OSPAR 2010). The ICES Working Group on Holistic Assessments of Regional Marine Ecosystems (WGHAME) has also recently updated its integrated assessment of the North Sea, and will continue its work as the Working Group on Integrated Assessments of the North Sea (WGINOSE).

Existing data sets are limited regarding their spatial and temporal scales, as well as the biodiversity components surveyed. Nevertheless, the ICES community is well placed to address the spatial patterns and temporal trends in the diversity of many marine taxa, for example through the various ecology groups, including: Working Group on Phytoplankton and Microbial Ecology (WGPME), Working Group on Zooplankton Ecology (WGZE), Benthic Ecology Working Group (BEWG), Working Group on Fish Ecology (WGFE), Working Group on Seabird Ecology (WGSE), Working Group on Marine Mammal Ecology (WGMME) and Working Group on Deepwater Ecology (WGDEC).

6.4.2 The roles of evolution, ecology and environment for biodiversity

The principal concepts structuring biodiversity were summarised in the previous WGBIODIV report (see Section 3.1 of ICES 2010a).

A wide range of marine scientists actively participate in the ICES community, and so utilising the skills and data from different disciplines will allow ICES expert groups to better understand how various environmental factors may affect 'biodiversity'.

6.5 Functional significance of biodiversity

6.5.1 The functional significance of genetic, species, population and ecosystem diversity

The significance of biodiversity for ecosystem functioning is largely linked to the resilience of ecosystems to perturbation, including functional redundancy forming an integral part of insurance from deterioration. This was discussed in Section 6.2.1.

6.5.2 Comparisons of system function and biodiversity

Important functions of an ecosystem include productivity, nutrient cycling, carbon sequestration/biological pump with a strong link to production, resistance to disease as well as habitat formation and complexity. For some of these functions substantial expertise exists within the ICES community including monitoring capacity (e.g. activities within WGPME, SGCBNS, SGPROD). It may be beneficial for ICES to facilitate collaborative approaches to better understand certain aspects of ecosystem function and its relationship with diversity.

6.6 Measuring 'biodiversity'

As measurements of biodiversity need to be considered on different levels, an overall assessment would need to be based on different scales (habitat, species and genetic scale). The overall concept of species diversity assessment was discussed by WGBIODIV in an earlier report (see Section 3.2 of ICES 2010a).

6.6.1 Measuring genetic diversity and the errors associated with these measurements

As an integral part of biodiversity, genetic diversity refers to the variation of genes within species. Within populations it includes gene diversity, heterozygosity and alleles per locus. Specifically gene diversity describes the proportion of polymorphic loci across the genome, heterozygosity means the mean number of individuals with polymorphic loci, whilst alleles per locus demonstrate genetic variability. The genetic structure of a population is defined by its gene pool's allele and genotype frequencies. Generally, genetic diversity within a species is the basis to maintain diversity among species, and *vice versa* (Lankau & Strauss 2007). Thus the adaptability of a species is based on its genetic adaptation potential.

Various scientists in the ICES community are experienced with population genetics, or have collaborated with population geneticists. The specialised laboratories required for molecular studies are typically associated with academic institutions, and some of the national fisheries institutes may not have in-house expertise.

Although genetic techniques are a useful tool to examine stock structure, genetic diversity etc., such studies are typically expensive and are hence typically applied to species of commercial, scientific or conservation interest. Within ICES, expertise exists in the Working Group on Application of Genetics in Fisheries and Mariculture (WGAGFM), but there is less coordination for advising on the genetic diversity in other groups. It should also be noted that advances in basic methodological research are essential to underpin genetic diversity assessments.

6.6.2 Measuring species diversity and the errors associated with these measurements

Various scientists in the ICES community are experienced with the identification of marine species and subsequent analyses of species diversity metrics. The main ecology expert groups (BEWG, WGFE etc.) are also well placed to undertake group-specific studies.

Some marine taxa are more problematic and relevant European experts may work in academic institutes and/or museums, where funds may not allow close liaison with the ICES community (e.g. for participating in expert groups). General data quality issues were discussed before (see ICES 2010a) and encompass catch rates related to gear types, trawl catches representing a fraction of a fish community, tows integrating over different discrete habitats, database errors, misidentification, distributional data relying on respective spatial coverage, seasonality of the data, data quality assurance and sampling effort (Borges *et al.* 2010).

6.6.3 Measuring habitat diversity and the errors associated with these measurements

Within ICES expertise exists for the provision of data on habitat types (through the BEWG, the Working Group on Marine Habitat Mapping (WGMHM), the Working Group on Ecosystem Effects of Fishing Activities (WGECO) and the Working Group on Deep-water Ecology (WGDEC)).

WGMHM have developed guidelines for data collection, based on a list of Recommended Operating Guidelines (ROGs) (see Section 6 of ICES 2010b). Based on the classification scheme implemented by EUNIS procedures, a facilitated classification was applied for Natura 2000 sites. Some gaps for analyses were identified by WGMHM including a lack of data, and this also applied for the grab sample data needed to ground-truth acoustic data (see Section 7 of ICES 2010b). Issues pertaining to the accuracy and confidence in habitat maps were also summarised and discussed by WGMHM (see Section 8 of ICES 2010b).

The provision of full biodiversity assessments for the ICES regions, emphasising different (temporal and spatial) scales, will depend on the quality of data provided and the integration achievable.

6.7 Projecting future changes in 'biodiversity'

Changes in the distributions of species and their movements and migrations due to environmental changes (e.g. Lewandowska & Sommer 2010) and changes in communities due to ocean acidification (e.g. coccolithophores, as well as effects on calcifying organisms) will probably affect wider ecosystem functioning. It has been suggested that diatoms will become major producers in coastal regions and that primary production will shift towards higher latitudes (Ducklow *et al.* 2010). Experimental work carried out in the North Atlantic open ocean suggests profound community shifts related to temperature, promoting the production of POC (Particulate Organic Matter) and an increase in dissolved CO_2 (Feng *et al.* 2009). It is highly likely that the described changes will have subsequent effects on food webs, multi-species interactions and nutrient cycling, with such processes strongly linked to biodiversity.

Within ICES, various Expert Groups are well suited to give advice on specific climate change queries, for example the Study Group on Working Hypotheses Regarding Effects of Climate Change (SGWRECC), the joint PICES/ICES Working Group on Forecasting Climate Change Impacts on Fish and Shellfish (WGFCCIFS) and the Study Group on Climate related Benthic processes in the North Sea (SGCBNS).

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7 Overview of the Census of Marine Life

7.1 Introduction

WGBIODIV were requested to "consider the results of the recently completed Census of Marine Life (CoML) project in the context of the ICES Science Plan".

The Census of Marine Life (CoML, http://www.coml.org) was a 10-year international program to examine the diversity, distribution and abundance of marine life (Costello *et al.*, 2010; McIntyre, 2010). The founders organized the Census around three grand questions: What did live in the oceans? What does live in the oceans? What will live in the oceans? They designed a program to explore the limits to knowledge of marine life in several areas around the world (Figure 7.1). The CoML involved about 2,700 scientists from more than 80 nations, and the results of the work have contributed towards many peer-reviewed publications (http://db.coml.org/comlrefbase/). The scientific results were reported on in October 2010.

Much of the work of the CoML was within well-defined projects, for example:

- Ocean Biogeographic Information System (OBIS): A web-based tool to provide access to geo-referenced information on the occurrence and distribution of marine species
- Pacific Ocean Shelf Tracking Project (POST): A program that applied electronic tagging technology to study the migratory routes of Pacific Salmon.
- Census of Coral Reefs Ecosystems (CReefs): An international census of coral reef ecosystems.
- Natural Geography in Shore Areas (NaGISA): An international project to examine biodiversity in inshore waters of <20 m depth.
- Gulf of Maine Area Program (GoMA): A project to examine patterns and processes in biodiversity in the Gulf of Maine.
- Continental Margin Ecosystems (COMARGE): An integrated program to examine patterns in the biodiversity of continental margins.
- Census of Diversity of Abyssal Marine Life (CeDAMar): A project to document species diversity on the deep-water abyssal plains.
- Mid-Atlantic Ridge Ecosystem Project (MAR-ECO): An international exploratory study of the fauna of the mid-Atlantic Ocean.
- Tagging of Pacific Predators (TOPP): A program to use electronic tags to study the migration patterns of large pelagic species in relation to oceanographic features.
- Census of Marine Zooplankton (CMarZ): A global biodiversity assessment of zooplankton.
- Global Census of Marine Life on Seamounts (CenSeam): A study of seamount ecosystems, in terms of biogeography, biodiversity and productivity.
- Biogeography of Deep-Water Chemosynthetic Ecosystems (ChEss): A study of the biogeography and processes of deep-water chemosynthetic ecosystems.
- Census of Antarctic Marine Life (CAML): A census of the Southern Ocean to better understand the biological diversity of this environment.

- Arctic Ocean Diversity (ArcOD): An international program to provide an inventory of the biodiversity in the Arctic (including that associated with sea ice, the water column and sea floor).
- International Census of Marine Microbes (ICoMM): A program to better coordinate what is known about microbes in the marine environment.
- History of Marine Animal Populations (HMAP): An inter-disciplinary research program to use archives of historical and environmental information to better understand the status of marine ecosystems prior to large-scale impacts from human activities.

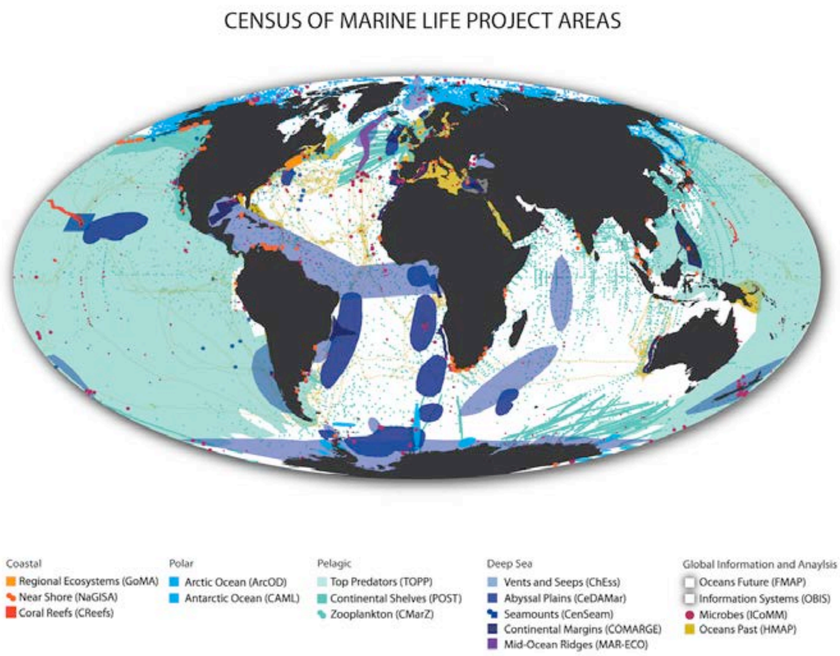

Figure 7.1. Overview of the Census of Marine Life Project areas (www.coml.org).

The Census encountered an unanticipated increase in the number of species, which are the currency of diversity. It upped the estimate of known marine species from about 230 000 to nearly 250 000. Among the millions of specimens collected in both familiar and seldom-explored waters, the Census found more than 6000 potentially new species and completed formal descriptions of more than 1200 of them. It found that some ostensibly rare species are common in some areas.

After all its work, the Census still could not reliably estimate the total number of species, the kinds of life, known and unknown, in the ocean. It could logically extrapolate to at least a million kinds of marine life that earn the rank of species and to tens or even hundreds of millions of kinds of microbes.

The European Census of Marine Life (EuroCoML) acted as the regional implementation committee for the CoML. Narayanaswamy *et al.* (2010a) have summarised some of the finding of the EuroCoML, with regional overviews of marine biodiversity produced for the Mediterranean Sea (Coll *et al.*, 2010; Danovaro *et al.*, 2010), the Atlantic coasts of western Europe (Narayanaswamy *et al.*, 2010b) and the Baltic Sea (Ojaveer *et al.*, 2010).

7.2 Outcomes of the CoML in relation to the ICES Science Plan

7.2.1 Patterns and Processes of the Ecosystems of the Northern Mid-Atlantic (MAR-ECO)

The major contributions from MAR-ECO project, studying animal life along the mid-Atlantic Ridge between Iceland and the Azores, were:

- Revised species inventories for the pelagic and demersal macro- and megafauna of the mid-ocean North Atlantic (Iceland-Azores section) based on MAR-ECO results and other sources.
- Revised range descriptions (and geo-referenced occurrence data in OBIS).
- Descriptions of new species from a number of taxa.
- New information on patterns of distribution and abundance at population and community level, with revised hypothesis that can form basis for studies in other geographical areas.
- Conceptual models and graphical outputs that synthesizes overall patterns.
- New information on genetic composition and population structure for deepwater species, especially fishes.
- New information on life history diversity of selected invertebrate taxa and fishes.

The information gained concerning exploited resources and biodiversity is of immediate value to global advisory authorities such as ICES and national research institutes, in turn benefiting regional management authorities such as the Northeast Atlantic Fisheries Commission (NEAFC) and the Oslo-Paris Commission (OSPAR). Enhanced knowledge on the mid-ocean ecosystems is critical to ensuring sustainable use and conservation of the oceanic environment and resources. Information derived from MAR-ECO was used as a significant scientific basis for recent decisions by NEAFC to close major sections of the mid-Atlantic Ridge to bottom fishing, and for OSPAR proposals to create mid-ocean MPAs.

MAR-ECO already expanded geographically. A spin-off project, which will last beyond the 2010 closure of the North Atlantic MAR-ECO, is underway in the South Atlantic.

Website: http://www.mar-eco.no/

Scientific output:

http://www.mar-eco.no/sci/mar-eco_publications/the_mar-eco_presentation_archive

7.2.2 Census of Diversity of Abyssal Marine Life (CeDAMar)

The goal of CeDAMar was to document actual species diversity of abyssal plains as a basis for global change research and for a better understanding of historical causes and actual ecological factors regulating biodiversity. The program focused on benthic, epibenthic and hyperbenthic organisms because of their high species-richness.

Most studies were carried out in the South Atlantic and the Southern Ocean, but the improved general knowledge about the abyssal plain will also be important in understanding such habitats in the ICES area.

Website: http://www.cedamar.org/

Publications: http://www.cedamar.org/Publications-resulted-from-CeDAMar

7.2.3 Continental Margin Ecosystems (CoMARGE)

The aims of CoMARGE were to: 1. Describe biodiversity patterns of benthic and bentho-demersal communities on continental margins, with a focus on multiple habitats and spatial scales; 2. Identify the contribution of environmental heterogeneities to these patterns. Led by French scientists, a number of field efforts included surveys and experiments along the European margin. This enhanced very significantly our knowledge on biodiversity on continental slopes, and also the temporal dynamics. A lot of baseline information and knowledge on processes were generated, relevant for advisory processes related to fisheries, biodiversity change, climate research, and deep-sea petroleum activity.

Website: http://www.ifremer.fr/comarge/en/index.html

Scientific output: http://www.ifremer.fr/comarge/en/Science1.html

7.2.4 Global Census of Marine Life on Seamounts (CenSeam)

Seamounts are often highly productive ecosystems, and may act as feeding grounds for fishes, marine mammals and seabirds. They are targeted for resource extraction such as fisheries and mining, but are ecologically vulnerable to such exploitation. At a global scale their biodiversity is poorly known with relatively few (< 200 of an estimated 100 000) seamounts having been studied in any detail.

CenSeam had two overarching priority themes; (1) What factors drive community composition and diversity on seamounts, including any differences between seamounts and other habitat types? (2) What are the impacts of human activities on seamount community structure and function? Several studies were conducted on North Atlantic Seamounts, and the project is highly relevant for ICES efforts to provide advice on seamount ecology, resources and human impacts.

Website: http://censeam.niwa.co.nz/

Publications: http://censeam.niwa.co.nz/science/censeam_science_publications

7.2.5 Biogeography of Deep-Water Chemosynthetic Ecosystems (ChESS)

ChESS was a global study of the biogeography of deep-water chemosynthetic ecosystems and the processes that drive them, and included studies of hydrothermal vents, seeps, whale falls etc., also in the North Atlantic. To ICES the insight gained is relevant to the advisory work on vents as vulnerable marine ecosystems mentioned in e.g. FAO guidelines. The project was led from NOC Southampton and had partners in many ICES countries.

Website: http://www.noc.soton.ac.uk/chess

Publications: http://www.noc.soton.ac.uk/chess/science/sci_publications.php

7.2.6 Census of Marine Zooplankton (CMarZ)

The CMarZ is a global, taxonomically comprehensive biodiversity assessment of animal plankton, including ~6800 described species in fifteen phyla. Several North Atlantic surveys were carried out, including full ocean depth sampling. CMarZ legacies include new baseline for detection of climate change; DNA technologies for rapid assessment of zooplankton species diversity for ocean observation and management. The CMarZ database contains species-level, specimen-based, geo-referenced entries; data and information are openly accessible via the CMarZ, CMarZ-Asia and PANGAEA websites, as well as the Ocean Biogeographical Information System (OBIS).

The major barcoding effort carried out is of great benefit to ICES work on zooplankton, but also for e.g. trophic studies beyond that group.

Website: http://www.cmarz.org/

Publications: http://www.cmarz.org/pubs_list.html

7.2.7 International Census of Marine Microbes (ICoMM)

International Census of Marine Microbes (ICoMM) has build a cyberinfrastructure to index and organize what is known about microbes, the world's smallest organisms, which account for an estimated 90% of biomass in oceans including an estimated 3.6 x 10^{30} microbial cells of untold diversity. The number of viral particles may be one hundred fold greater. This project is relevant to ICES efforts on microbiology and ecosystem processes (biotic and abiotic).

Website: http://icomm.mbl.edu/

7.2.8 Pacific Ocean Shelf Tracking (POST)

The POST Project was designed to develop and promote the application of acoustic tagging technology to study the life history of Pacific salmon. A major area of focus for POST involved the development of a permanent continental-scale telemetry system. POST's array sits on the seabed of the continental shelf and upstream in several major rivers, and is used to monitor the movements of not only salmon, but many other types of marine animals along the shelf. The technology is relevant for the study of anadromous fish resources in the ICES area.

Website: http://www.postcoml.org/

Publications:
http://www.postcoml.org/page.php?section=community&page=publications

7.2.9 Tagging of Pacific Predators (TOPP)

The TOPP program used electronic tagging technologies to study the migration patterns of large open-ocean animals and the oceanographic factors controlling these patterns. The technology has documented in an unprecedented manner the long-range migrations of top predators, and the technology has great potential for corresponding studies in the ICES area.

Website: http://www.topp.org

Publications: http://www.topp.org/topp_census#pubs

7.2.10 History of Marine Animal Populations (HMAP)

The HMAP project was an interdisciplinary research program using historical and environmental archives to analyze marine population data before and after human impacts on the ocean became significant. HMAP had a case study approach. The case studies were generally regional in scope and focused on a few species of commercial importance or habitat and biodiversity changes. Individual studies were selected on the basis that the ecosystem has been subject to fishing and that there existed sufficient historical data on catches and harvesting effort. Examples, many of which are relevant for the ICES area:

- Northwest Atlantic (Gulf of Maine, Newfoundland-Grand Banks, Greenland cod fisheries);

- Southwest Pacific (Southeast Australian Shelf and Slope fisheries, New Zealand Shelf fisheries);
- White and Barents Seas (Russian and Norwegian herring, salmon and cod fisheries, and Atlantic walrus hunting);
- Norwegian, North and Baltic Seas (Multinational cod, herring and plaice fisheries);
- Southwest African Shelf (Clupeid fisheries in a continental boundary current system);
- Worldwide Whaling (Historical whaling in all oceans);
- Caribbean communities (Impact of the removal of large predators).

Many HMAP projects were interpreting changes in marine populations over the past 500–2000 years, which provides researchers of current and future conditions with a baseline that extends back long before the advent of modern technology, or before significant human impacts on the ecosystem.

Website:
http://www.hull.ac.uk/hmap/hmapcoml.org/History%20of%20Marine%20Animal%20Populations.swf

Publications: http://hmapcoml.org/publications/

7.2.11 Future of Marine Animal Populations (FMAP)

The Future of Marine Animal Populations (FMAP) was a network of scientists within the Census of Marine Life trying to understand the past, present and future of marine life. FMAP's mission was to describe and synthesize globally changing patterns of species abundance, distribution, and diversity, and to model the effects of fishing, climate change and other key variables on those patterns. This work was done across ocean realms and with an emphasis on understanding past changes and predicting future scenarios. FMAP had a strong emphasis on statistical modelling of patterns derived from biological data. The project's focus was on data synthesis, often by means of meta-analysis, which is the formal integration of many data sets to answer a common question. Questions were asked about the status and changes in diversity, abundance and distribution of marine animals, such as:

- What are the global patterns of marine biodiversity?
- What are the major drivers explaining diversity patterns and changes?
- What is the total number of species in the ocean (known and unknown)?
- How has the abundance of major species groups changed over time?
- What are the ecosystem consequences of fishing and climate change?
- How is the distribution of animals in the ocean changing?
- How is the movement of animals determined by their behaviour and environment?

Website: http://www.fmap.ca/

Publications: http://www.fmap.ca/publications.php

7.2.12 Ocean Biogeographic Information System (OBIS)

OBIS is a web-based provider of global geo-referenced information on marine species, with online tools for visualizing relationships among species and their environment. Data are volunteered by many sources and all CoML projects, and OBIS has now become housed by IOC-UNESCO. It provides expert data on marine species and currently contains nearly 30 million geo-referenced species records from more than 800 databases (numbers of September 2010). This is a new source of data end entry to relevant literature for ICES scientists and expert groups, and actively providing ICES datasets to OBIS would complement the data already there, enhance the quality and extend spatial coverage of OBIS output.

Website and portal: http://www.iobis.org/

7.3 CoML significance for ICES

The CoML legacy is new baseline information on biodiversity across ocean realms, and new networks of scientists and students of biodiversity and ecosystem processes. The ICES science plan emphasises the following items, and CoML has added significant new information and competence relevant to all:

- Understanding ecosystem functioning
- Understanding of interactions of human activities with ecosystems
- Development of options for sustainable use of ecosystems
- Enhanced research coordination in the North Atlantic
- Enhanced science capacity

For the North Atlantic and Arctic, the enhanced competence and networks studying deep-sea biology and processes, both on ocean margins, ridges, chemosynthetic systems, and in shelf waters should be emphasised. The historical ecology as a field of research has gained momentum and is highly relevant to ICES expert groups such as SGHIST (ICES 2010). All these areas had previously a rather limited emphasis in the ICES community. The challenge may now be to involve relevant networks, and especially people from academia (including natural history museums), in the ICES activities.

CoML further provides the ICES community with a readily accessible global overview of issues, new science, contacts and activity. This may enhance ICES global activity and perspective.

OBIS is a tool with significant potential for use in ICES, but it is still under development and in need of improved support in terms of software, quality control and more complete datasets.

Given that many of the exploratory surveys undertaken during the CoML have enabled many new species to be described, including from those undertaken in the ICES area, there is clearly still much to be learnt about marine biodiversity. Further exploratory surveys and expeditions to better understand certain taxa, habitats and regions could usefully be undertaken, and could even form an extension of the field work currently supported (in part) by the Data Collection Framework, even if on an occasional basis.

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Annex 2: WGBIODIV draft resolution for the next meeting

The **Working Group on Biodiversity Science** (WGBIODIV), chaired by Jim Ellis, UK, will meet at VENUE [to be confirmed: either ICES Headquarters or Hamburg or Nantes], 30 January –3 February 2012 to:

a) Provide a critique of the current proposed metrics in support of MSFD Descriptor 1 (EC Decision Document of 01/09/2010), including:

(i) undertaking case studies of species/habitat biodiversity metrics/indicators proposed by individual Member States, to assess their usefulness for informing on the maintenance of 'biodiversity' as defined by the CBD,

(ii) identifying other potential biodiversity metrics that could usefully inform on other aspects of biodiversity that may not be covered by species-specific metrics,

(iii) consider how the different metrics applied in the case studies examined might be used to derive a more regional scale assessment of biodiversity status,

(iv) comment on the appropriate geographic scales for ensuring trans-boundary species/habitats are assessed at biologically-meaningful scales,

WGBIODIV will report by 15 March 2012 (via SSGEF) for the attention of SCICOM.

Supporting information

Priority	High. The work of the Group is essential if ICES is to progress with making biodiversity an integral part of ICES work, especially given the recent Marine Strategy Framework Directive.
Scientific justification	Biodiversity is explicitly addressed in the ICES Science Plan 2009-13 as follows: biodiversity can be considered at a number of scales in marine ecosystems – from the genetic and population level, through the species level up to the community level. It may be a key element of the capacity of an ecosystem to absorb disturbance without shifting to another regime – its resilience. It is generally accepted that relatively high (i.e. intact or non-reduced) biodiversity operating at each level confers plasticity and resilience. These are essential attributes under conditions of change due to natural and anthropogenic factors and thereby indicators of a healthy ecosystem. The study of the relative resilience of shelf seas exploited ecosystems through a comparative approach will provide knowledge and understanding of biodiversity which will be of importance to several research topics. WGBIODIV will address the key scientific issues in close cooperation with the concomitant Strategic Initiative led by SSGSUE.
Resource requirements	No specific resource requirements beyond the need for members to prepare for and participate in the meeting.
Participants	Expertise from all areas of the marine benthic and pelagic food web components. Participation is sought from ICES countries and by scientists both from disciplines and scientific circles not normally represented at ICES.
Secretariat facilities	Not exceeding the usual requirement
Financial	None specific.
Linkages to advisory committees	ACOM.

Linkages to other committees or groups	The work of the group can be linked to some of the work of the various ecology expert groups (e.g. BEWG, WGFE, WGZE etc.) and survey groups (e.g. WGBEAM, IBTSWG)
Linkages to other organizations	CBD, IMoSEB, OSPAR, HELCOM