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30 January – 3 February 2012

Nantes, France

ICES International Council for the Exploration of the Sea

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Executive summary

Biodiversity is an increasingly important element of ICES' work, and was identified in the ICES Science Plan as a research topic of strategic importance. The 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 likely increase in the future. Although a range of ICES Expert Groups are involved in various aspects of marine biodiversity, WGBIODIV aims to provide the ICES community with an improved capacity to coordinate, integrate and synthesise biodiversity information.

Progress with the development of biodiversity indicators by EC nations in the ICES area was reviewed, although many nations were still consulting on potential metrics and indicators at the time of the meeting (Section 2). This section included a case study illustrating potential species level indicators and targets for monitoring demersal fish biodiversity in the North Sea, which can be utilised with existing trawl survey data. Potential methods for estimating the distributional range and patchiness of species (or species groups) were also discussed and examples of these approaches given for fish in the Bay of Biscay and Barents Sea. Comments and examples for other potential metrics, including for indicators for threatened species, the application of the Large Fish Indicator (LFI), the utility of the mean maximum length of all fish observed in trawl surveys, and the appropriate geographical scales for case study species, were also given.

The role of 'biodiversity' in supporting 'ecosystem functions' is an area of increasing scientific interest, as biodiversity is thought to enhance the integrity and resilience of ecosystems. Ecosystems provides a wide variety of functions, including for supporting services (e.g. habitat formation and primary production) and regulating services (e.g. nutrient cycling and energy flow, oxygenation and gas exchange, decomposition and the biological regulation of populations). Although there has been a range of studies to link 'biodiversity' to 'ecosystem function' in terrestrial and freshwater ecosystems, there have been comparatively few studies evaluating biodiversity indicators that are tightly linked to ecosystem function in marine systems, as noted in Section 3, and further studies on this topic are required.

WGBIODIV were also asked to review on methods to integrate biological data from disparate sources and for different components of the ecosystem to better describe biodiversity hotspots. An improved knowledge of biodiversity hotspots in the ICES area is required, as this may help inform on monitoring programmes and analyses of field data, and may also have a role in spatial management, as such hotspots are often promoted as a cost-effective method of protecting biodiversity. Although the term 'biodiversity hotspot' is used widely, there is surprisingly little consensus on its actual definition. For the purposes of the present report, it was assumed that 'biodiversity hotspots' may represent sites or regions where there is (a) an increased biomass, density, and/or species richness of one or multiple taxonomic groups; (b) a high productivity; (c) a high level of endemism; (d) a concentration of rarer species and habitats; or (e) a high concentration of species that are aggregating for ecologically-important functions. A brief overview of biodiversity hotspots is given in Section 4, with case studies providing illustrating various issues, including gear catchability, methods of integrating data and modelling fish distributions to better identify abundance hotspots.

The final part of the report (Section 5) provides examples of how new technologies can support improved biodiversity monitoring and assessment. Despite the various new technologies that can support biodiversity studies, biodiversity monitoring has, through its taxonomic component, a strong conservative aspect that needs to be retained when trying to achieve reliable and comparable results. It should also be recognised that long-term biodiversity monitoring requires commitment to long-term monitoring programmes and standardised data collection. Supplementary field methods and special technologies, if and when incorporated into existing monitoring programmes, should typically be additional studies, and not compromise long-term datasets.

1 Introduction

1.1 Background

The Study Group on Biodiversity Science (SGBIODIV) first met in 2007 in Belgium (ICES, 2007) and reported on the possible role of ICES in terms of biodiversity science. The following year, SGBIODIV reviewed current and emerging marine biodiversity initiatives, and provided an overview of how various ICES Expert Groups contributed to biodiversity science (ICES, 2008). In 2009, SGBIODIV met in Germany, and provided options for the better integration of biodiversity science across the ICES science and advisory community (ICES, 2009). During this meeting, 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".

The group, re-named the Working Group on Biodiversity Science (WGBIODIV), met in Lisbon, Portugal the following year, and the report 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 (ICES 2010). In terms of developing indicators of biodiversity, WGBIODIV also 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).

In 2011, WGBIODIV met at ICES headquarters, and summarised some of the potential methods for examining the diversity of multiple groups and some of the approaches to mapping various facets of biodiversity information (including comments on the spatial distribution of distinct faunal assemblages in certain parts of the ICES area) and undertook case-study examples of potential biodiversity indicators (ICES, 2011). WGBIODIV also gave further examples of how survey data may inform on biodiversity metrics, with emphasis on potential pitfalls (e.g. gear selection, site selection, density of sampling stations, sample replication, catch processing, taxonomic resolution, and data filtering and standardisation); and provided a brief summary of the Census of Marine Life (CoML).

In terms of policy, the two drivers for the assessment of biodiversity are the Convention on Biological Diversity (CBD) and, for European nations, 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" (European Commission, 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 Commission has also given examples on criteria and potential indicators for assessing biodiversity (European Commission, 2010), and how such criteria may be applied across the OSPAR area has been discussed (OSPAR Commission, 2011a,b).

1.2 Definitions

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".

1.3 Terms of Reference

The **Working Group on Biodiversity Science** (WGBIODIV), chaired by Jim Ellis, UK, will meet in Nantes, France, 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 transboundary species/habitats are assessed at biologically-meaningful scales.

b) Review and report on methods to integrate biological data from disparate sources and for different components of the ecosystem to better describe biodiversity hotspots;

c) Review and report on new technologies that can support improved biodiversity monitoring and assessment;

d) Review and report on existing indicators of biodiversity that are linked to predictable changes in ecosystem function and/or to develop, assess and report on the feasibility and performance of such indicators.

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

1.4 Participants

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

Oscar Bos	Netherlands
Anik Brind'Amour	France
Jan Ekebom	Finland
Jim Ellis	UK (England & Wales)
Simon Greenstreet	UK (Scotland)
Edda Johannessen	Norway
Pascal Laffargue	France
Nikolaus Probst	Germany
Isabelle Rombouts	France
Heye Rumohr	Germany
Verena Trenkel	France
*Ángel Borja	Spain (Basque Country)
*Steven Degraer	Belgium
*Leonie Dransfeld	Ireland
*Miriam Guerra	Portugal
*Henn Ojaveer	Estonia

1.5 Summary of Working Documents and presentations

Oscar Bos gave a presentation on "Draft OSPAR's MSFD Advice Manual on Biodiversity approaches to determining GES, setting of environmental targets and selecting indicators for MSFD Descriptors 1, 2, 4 and 6", which gave a summary of recent work undertaken by OSPAR's Biodiversity Committee (BDC) and Intersessional Correspondence Group for the Coordination of Biodiversity Assessment and Monitoring (ICG-COBAM), which has recently held workshops to discuss approaches for setting targets and baselines for biodiversity indicators (Utrecht, November 2010), and to compare nationally selected indicators and targets, and to identify common indicators (Amsterdam, November 2011).

1.6 Format of Report

The work undertaken by WGBIODIV on biodiversity indicators to address ToRs (a) and (d) is detailed in Sections 2 and 3 of this report. Section 4 provides a summary overview of methods to integrate biological data from disparate sources and for different components of the ecosystem to better describe biodiversity hotspots; and Section 5 provides a brief review of some of the new technologies that can support improved biodiversity monitoring and assessment.

Given the expertise of the participating scientists, the information provided in this report focuses on species-based analyses of biodiversity, with less information of relevance to genetic and habitat diversity.

1.7 References

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Cochrane, S.K.J., Connor, D.W., Nilsson, P. ,Mitchell, I., Reker, J., Franco, J., Valavanis, V., Moncheva, S., Ekebom, J., Nygaard, K., Serrao Santos, R., Naberhaus, I., Packeiser, T., van de Bund, W. and Cardoso, A.C. 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.

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

ICES. 2011. Report of the Working Group on Biodiversity Science (WGBIODIV), 21–25 February 2011, ICES Headquarters, Copenhagen, Denmark. ICES CM 2011/SSGEF:02. 94 pp.

OSPAR Commission. 2011a. Report of the OSPAR/MSFD workshop on approaches to determining GES for biodiversity. OSPAR Commission Biodiversity Series, Publication Number: 553/2011, 56 pp.

OSPAR Commission. 2011b. Identification of ecological monitoring parameters to assess Good Environmental Status of marine waters. An inventory in all OSPAR Contracting Parties that implement the MSFD. OSPAR Commission Biodiversity Series, Publication Number: 554/2011, 50 pp.

2 Progress on biodiversity indicators for the Marine Strategy Framework Directive

2.1 Introduction

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…" (European Commission (EC), 2008). The directive aims to achieve Good Environmental Status (GES) by 2020 and its major programme is biodiversity-related.

Of the eleven 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).

2.2 Brief overview of EC decision document on types of biodiversity indicators

The EC published supporting information on potential criteria and methodological standards on good environmental status of marine waters (EC, 2010). With regards Descriptor 1, the guidance highlighted that:

- "Assessment is required at several ecological levels: ecosystems, habitats (including their associated communities, in the sense of biotopes) and species"
- "To address the broad scope of the descriptor, it is necessary … to prioritise among biodiversity features at the level of species, habitats and ecosystems. This enables the identification of those biodiversity features and those areas where impacts and threats arise and also supports the identification of appropriate indicators among the selected criteria…"
- "For each region, sub-region or subdivision, taking into account the different species and communities … contained in (Table 1 of Annex III to Directive 2008/56/EC), it is necessary to draw up a set of relevant species and functional groups …. The three criteria for the assessment of any species are species distribution, population size and population condition."

Suggested criteria for assessing the biodiversity of species and habitats, as given in EC (2010) are summarised in Table 2.1.

Table 2.1. Levels, criteria and indicator types proposed for Descriptor 1 "Biological diversity is maintained" under the MSFD (Adapted from EC, 2010). Light grey cells illustrate habitat-level indicators, or indicators explicitly directed towards benthic invertebrates and other sessile species, and so considered here to be of little relevance to assemblages of motile animals. Intermediate grey cells indicate the ecosystem-level indicator addressed elsewhere. Dark grey cell indicates the single species-level indicator for which insufficient information is available to develop a fish assemblage biodiversity indicator.

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)

2.3 National progress in developing indicators of marine biodiversity

Most Member States have made progress in developing biodiversity indicators, but at the time of the WGBIODIV meeting, several Member States were in the process of consultations, and so it was not possible to evaluate specific proposals in line with the ToR.

2.3.1 Belgium

Belgium aims to develop operational indicators and targets that will take into account the MSFD, the accompanying Commission Decision, the Habitats & Bird Directive as well as other legal and conventional commitments, such as OSPAR. The first list of indicators is being consulted on and will provide a tool to determine GES during the first MSFD cycle. This list will be refined and developed as new data and knowledge becomes available to inform on the second MSFD cycle.

2.3.2 Denmark

No information was available during the meeting.

2.3.3 Estonia

Estonia has produced a draft Initial Assessment, according to the MSFD Article 8, and this document will be finalized shortly. Discussions on the proposed indicators will then be held.

2.3.4 Finland

No specific information was available during the meeting, although progress made by HELCOM in relation to MSFD assessments are summarised in Section 2.5.

2.3.5 France

France is responsible to define indicators for part of the English Channel, a small part of the Celtic Sea, Bay of Biscay and parts of the Mediterranean Sea (primarily the Gulf of Lions).

In France, a qualitative approach to describe GES for Descriptor 1 has been now finalized. Considering the current gaps of knowledge in the state of many biological components and notably their potential resilience (capacity to recover from impacts of anthropogenic pressures), it seemed inappropriate to adopt a quantitative approach at this stage and consequently, no quantitative targets have been set to date.

Nevertheless, considerable progress has been made in developing common methodologies, e.g. setting of parameters and selection procedures of biological components (functional group of species and habitat types) in collaboration with neighbouring Member States (cf. Ospar/ICG-COBAM Advice Manual). As a result, a common set of criteria (listed and common species, functional importance and high diversity) was adopted to establish a list of species and habitats in France. However, these lists remain works in progress and will be further compared with those of other Member States to identify commonalities and facilitate compatibility for establishing monitoring programmes (according to the future task for the period 2012–2014).

Using the available data in French subregions, both species and community metrics will be calculated to derive biodiversity indicators. Furthermore, by combining these metrics, the importance of diversity in the functioning of trophic food webs could be evaluated via a bottom-up and a top-down approach, in accordance with Descriptor 4 indicators.

However, before a quantitative assessment for GES in France can be achieved, methodological improvements are needed to further develop and combine appropriate metrics to evaluate and measure potential changes in biodiversity. In particular, additional work will focus on the relevant spatial scale over which species and community metrics should be calculated and for this purpose, a "case by case" approach will be adopted. It is highly likely that an integrative approach over different scales in relation to pressure indicators will be used to facilitate the interpretation of the D1 indicators for evaluating GES in space and time.

2.3.6 Germany

Germany has defined potential indicators with GES-targets for the North Sea and Baltic Sea. The documents are only available in German (see www.meeresschutz.info). In the selection of indicators, Germany has focused on already existing information from implemented marine policies such as the WFD and HD as well as international organisations such as ASCOBANS, HELCOM, ICES and OSPAR.

The suggestions of the German MSFD indicators are in most cases preliminary and not operational at the current state (as of January 2012). The targets for GES are mostly quantitative and the rational for their selection is mostly based on the aforementioned policies and committees. For many descriptors a final list of relevant species and/or communities is still under development.

2.3.7 Ireland

In Ireland, there has been no formal adoption of MSFD indicators, metric calculations and target setting. However, Ireland is working within the OSPAR and ICES framework to develop and coordinate their MSFD related biodiversity indicators.

Selection of species: Preliminary list of mobile species (seabirds, marine mammals, fish and cephalopods) for the Celtic Sea MSFD subregion have been compiled and submitted to OSPAR. Reptiles (marine turtles) were not included on the list due to a lack of monitoring data. Selection criteria were based on proposed OSPAR criteria as presented to ICG COBAM 2011, including:

- their abundance and distribution (i.e. naturally predominant species as well as species that are predominant as an effect of human activities should be included);
- their sensitivity towards specific human activities;
- their suitability for the respective indicators and descriptors of the EU COM decision;
- the practicability (including cost effectiveness) to monitor them;
- their inclusion in existing monitoring programmes and time-series data;
- their association with specific habitats.

To explore data availability, species were mapped against the different D1 indicators as given by the Commission's decision document. As the currently proposed D4 indicators of this document also cover population abundance and productivity, these were considered to be covered by the data requirements of D1 indicators, as long as the specific D4 criteria were also addressed (e.g. for species at the top of food web, species with fast turnover rates).

For demersal teleosts and elasmobranchs, data series of the Irish groundfish survey were used. This survey is part of International Bottom Trawl Survey (IBTS). Species were selected on the basis of commonness, vulnerability and the other above listed criteria. The list covers a range of commercial and non-commercial species. Commercial species were included in D1 as they are important components of the fish community but also because the indicators of D1 cover some aspects (e.g. distributional range and pattern) that D3 does not cover.

The pelagic fish species on the species list are predominantly commercial as these are the most abundant pelagic species in the Celtic Seas ecosystem for which monitoring data exists. Mesopelagic species were not included. Although these are very abundant and important components of the pelagic ecosystem, they do not have dedicated surveys and monitoring their status would be problematic.

Important species that have been selected according to the selection criteria but lack monitoring data were identified and flagged for consideration of future monitoring programmes, or for alternative indicator selection. This is the case for pelagic sharks, for example, for which monitoring is problematic. Bycatch numbers could be considered as an alternative to abundance estimates. The selection of deepwater teleosts and elasmobranchs was based on the Irish deepwater survey programme; however the continuation of this monitoring programme is subject to future funding and will determine if the species can be included in the assessment.

Selection of indicators and metric development: The selection process for indicators under D1 focussed strongly on the Commission's Decision Document (EC, 2010) and the OSPAR biodiversity advice manual (OSPAR, 2011c) which proposes a core set of indicators for which there is high level of regional agreement. For selected fish and cephalopod species, initial indicators on the species level are distributional range, distributional pattern within the latter, and population abundance and biomass. For population structure, the same indicators are considered as under D3, namely the proportion of fish larger than the mean size of first sexual maturation and the 95% percentile of the length distribution. The applicability of three fish community indicators are currently being explored for Indicator 1.6.1 (Condition of the typical species and communities), namely the large fish indicator (LFI), the conservation status of fish species (CSF) and the mean maximum length of the demersal fish community. The LFI is also considered for indicator 4.2.1 in the foodweb descriptor. This is in line with recommendations of the OSPAR workshop on biodiversity indicators (OSPAR, 2011a) and the OSPAR advice manual on biodiversity descriptors (OSPAR, 2011c).

For all indicator time series, different methods to detect significant trends and/or deviation from the baseline are currently being explored. These include the intersection – union tests for characterising recent changes in the indicator time series (Trenkel and Rochet, 2009) and the mean of recent years compared against either the total mean of the time-series or a historic mean. All three methods are explored on indicator time series for commercial fish species for Descriptor 3 in ICES (2012).

2.3.8 Latvia

No specific information was available during the meeting, although progress made by HELCOM in relation to MSFD assessments are summarised in Section 2.5.

2.3.9 Lithuania

No specific information was available during the meeting, although progress made by HELCOM in relation to MSFD assessments are summarised in Section 2.5.

2.3.10 The Netherlands

The indicators for the GES, including targets, have been compiled in scientific background reports by IMARES/Deltares and are currently put in the policy documents of the MSFD that will be submitted to the EC in July 2012. The Netherlands has data for fish, benthos, seabirds, marine mammals and metrics will focus on these groups and includes both new indicator groups such as long-lived benthic species, and EcoQos developed by OSPAR.

2.3.11 Poland

No specific information was available during the meeting, although progress made by HELCOM in relation to MSFD assessments are summarised in Section 2.5.

2.3.12 Portugal

Portugal has identified potential quantitative and qualitative data sources (either from IPIMAR or from bibliographic sources) and ecosystem components (including macroalgae, phytoplankton, zooplankton, macrozoobenthos, megabenthos, fishes, marine mammals, seabirds and sea turtles). The development of Portuguese indicators will focus on rare and ecologically significant species and endangered habitats according to the 2010/477/EU Directive.

2.3.13 Spain

In the case of Spain, the Basque Country proposed an integrative assessment within the WSFD (Borja *et al.*, 2011). Regarding biodiversity, these authors proposed using integrative tools, such as the biodiversity valuation approach, in assessing biodiversity within the MSFD. The valuation of biodiversity is in response to the continuing requests of policy-makers and marine managers for reliable and meaningful biological baseline maps, to be able to make well-deliberated selections, concerning the sustainable use and conservation of the marine environment. Biodiversity valuation maps aim to compile all available biological and ecological information for a selected study area and allocate an integrated intrinsic biological value to the sub-zones (Derous, 2007; Derous *et al.*, 2007).

For the Basque Country, data on zooplankton, macroalgae, macroinvertebrates, demersal fish, sea mammals and seabirds, for the period 2003–2009, and for the whole Basque continental shelf, were collated. The integrative biodiversity value of the Basque continental shelf was based on the methods of Pascual *et al.* (2011), from which it is possible to integrate the biodiversity valuation into a unique value for the whole of the Basque continental shelf (Borja *et al.*, 2011); this is a similar approach to the Ecological Quality Ratio (EQR), within the Water Framework Directive. In this particular case, reference conditions for high values do not exist; as such environmental targets, as demanded by the MSFD, rather than reference conditions, can be used (see Borja *et al.*, 2012). Such targets can guide progress towards achieving good environmental status; for biodiversity, those for 'high' value can be adopted.

No specific information for the rest of Spain was available during the meeting, as potential metrics and indicators were being consulted on at the time of the meeting.

2.3.14 Sweden

No specific information was available during the meeting, although progress made by HELCOM in relation to MSFD assessments are summarised in Section 2.5.

2.3.15 United Kingdom

No specific information was available during the meeting, as potential metrics and indicators were being consulted on at the time of the meeting. However, examples of some of the types of indicators being explored are given below.

2.4 Comments on metrics proposed for supporting MSFD Descriptor 1

2.4.1 Introduction

Scientists from different disciplines may be proponents of different aspects of single-species or multispecies-metrics, whether the classical diversity metrics, taxonomic distinctness, functional diversity, or size-based metrics. Whereas appropriate data collection may allow multiple approaches to be used, there can be issues of redundancy when multiple metrics are presented.

There has been much discussion in the scientific literature of particular taxa acting as 'surrogates' of wider biodiversity. Some taxa may be more responsive to particular anthropogenic or environmental drivers than others and it has been found that assemblages of different taxonomic groups may respond differently to environmental gradients (see Heino 2010). Hence, there may be little congruence in the species richness of differing taxa in various habitats, and the utility of indicator taxa is limited in some respects (Heino *et al.* 2005). Other studies have highlighted that some species within the assemblage may be useful indicators of environmental conditions, such as using ichthyoplankton data to inform on the broad type of oceanic conditions (Brodeur *et al.* 2008).

One of the main problems with 'biodiversity indicators' is that there is no single, measurable target, as biodiversity encompasses so many factors, including species richness across a wide range of taxa; the genetic diversity within each species; the structure, composition and extent of habitats and assemblages; functional diversity; ecosystem services and status of threatened species, as well as public perceptions to any intrinsic and/or cultural value of particular species and habitats (Mace & Baillie 2007). Hence, other organisations have identified multiple metrics, which may be used in the form of 'composite indicators'.

OSPAR has compiled information on biodiversity metrics proposed by different member states through a number of workshops in a number of summary tables in their MSFD advice manual (OSPAR, 2011c). The proposed metrics were not reviewed by WGBIODIV, as most of them were still in a conceptual phase. Only after the metrics have been submitted to the EC by the Member States (July 2012) will it be possible to examine them in detail.

Here we provide some comments on a range of generic issues related to metrics of 'sensitive species' and maximum length, as well as the integration of indicators by providing case studies (see below).

2.4.2 Indicators for threatened species

The MSFD should "support the strong position taken by the Community, in the context of the Convention on Biological Diversity, on halting biodiversity loss" (EC, 2008), and within Annex III of the Directive, the Indicative lists of characteristics, pressures and impacts highlights that biological features includes "a description of the population dynamics, natural and actual range and status of other species occurring in the marine region or subregion which are the subject of Community legislation or international agreements".

EC (2010) acknowledged that given the "broad scope of the descriptor (Descriptor 1), it is necessary … to prioritise among biodiversity features at the level of species, habitats and ecosystems. This enables the identification of those biodiversity features and those areas where impacts and threats arise and also supports the identification of appropriate indicators among the selected criteria…".

Hence, there is a need for Member States to address 'threatened species'. Examples of the types of fish that may be listed as of 'conservation interest' under various nature conservation conventions and listings were given in a previous report (see Table 3.1 of ICES 2011). This report also highlighted some of the problems associated with identifying species on the basis of 'red lists of threatened species', including that for many cases, the listed species are essentially charismatic, flagship species that have been 'championed' by individuals or organisations, and analyses to rank all taxa (within specified marine groups) by order of sensitivity or threat status have not been undertaken.

It is also important to note that some species which are "the subject of Community legislation or international agreements" are often not sampled effectively in existing surveys, and there can be various reasons for this. Some species (e.g. *Squatina squatina*, *Rostroraja alba*) may now be too rare in former habitat, that they are no longer encountered in surveys. Some of the other species are not effectively sampled by existing surveys, including many of the faster-moving or large pelagic fish (e.g. *Cetorhinus maximus*). In some instances, the main habitat may be outside the primary survey area of existing surveys (whether in more coastal waters, or in deeper waters), and so an expansion of survey area (without compromising the existing survey) would be required if more reliable data were to be collected.

Although some proponents have suggested that the broad-scale biodiversity of a nation or region may be informed by the conservation status of relevant species, such an approach is not necessarily informative, and Quayle & Ramsay (2005) suggested that "Changes in conservation lists are thought to be of limited utility as biodiversity indicators because they may reflect changes in human knowledge of species status better than they indicate actual change in status itself".

2.4.3 Case study: species level indicator and targets for North Sea demersal fish

This case study examined the selection of species level indicators for the North Sea demersal fish assemblage and how to set trend-based targets under D1 (Maintenance of Biodiversity) of the MSFD.

Genetic structure and diversity has only been examined in a few fish populations, often involving commercial species (Mork *et al.*, 1985; Giæver and Forthun 1999; Shaw *et al.*, 1999; Nesbø *et al.*, 2000; Hutchinson *et al.*, 2001; Nielsen *et al.*, 2009). Although this case study (and the wider report) does not consider genetic diversity, this does not mean that genetic losses have not occurred, but rather it is a consequence of the lack of available data. Furthermore, genetic information is relevant to the development of indicator 3.3.4 (size at first sexual maturation, which may reflect the extent of undesirable genetic effects of exploitation) for Descriptor 3 "Populations of all commercially exploited fish and shellfish". No demersal fish metrics to support this indicator were thought to be feasible. Ecosystem-level indicators and targets are considered elsewhere in the reports of WGBIODIV.

Groundfish surveys have been carried out for decades in support of fisheries management (Heessen, 1996; Heessen and Daan, 1996). The abundance-at-length data generated by these surveys are ideal for determining metrics to populate these four species-level indicators for fish assemblages (Trenkel and Cotter, 2009).

The Q1 IBTS is the longest-running groundfish survey still in operation and covers most of the Greater North Sea; it is therefore the data set used to calculate the North Sea Large Fish Indicator (LFI) on which the current OSPAR EcoQO is based (Greenstreet *et al.*, 2011, 2012; Heslenfeld and Enserink, 2008). Here the same data set is used to exemplify its potential in supporting MSFD D1 indicators.

A second groundfish survey also operates in the North Sea, the Q3 IBTS. While Q3 IBTS data are available from 1991 onwards, a major redesign of the survey occurred in 1998. Greenstreet (submitted) concluded that only the data from 1998 onwards were suitable for assessing changes in North Sea demersal fish biodiversity, rendering this time series too short for this case study. However, changes in species richness observed from the Q3 IBTS were similar to those determined from the Q1 IBTS data and in the years to come, as nations move forward with implementing the MSFD, the Q3 IBTS should still prove to be an extremely useful monitoring programme, serving to corroborate the trends observed from the Q1 IBTS and offering a degree of seasonal information.

Biodiversity-related metrics are notoriously affected by variation in sampling regime (Greenstreet and Piet, 2008). Groundfish survey data therefore often require a degree of standardisation prior to any analysis (Jouffre *et al.*, 2010). Consequently, Greenstreet (submitted) devised a "standard survey area" for the Q1 IBTS so that only data from an area more or less fixed in space were analysed each year. The extent of this area can be quantified in terms of the number of ICES statistical rectangles surveyed each year and the proportion of these rectangles in which particular species were recorded can be used as an indication of variation in their distributional range (Table 2.1; indicator 1.1.1). Indicators of species abundance and/or biomass (Table 2.1; indicator 1.2.1) can obviously be generated directly from the IBTS data. For the present study, the mean:variance ratio of the ICES statistical rectangle species abundance or biomass data (positive occurrence only) for the rectangles in which each species was recorded has been used as an indicator to monitor variation in distributional pattern within the occupied range (Table 2.1; indicator 1.2.1). Finally, spawning stock biomass has been a key indicator used to assess the state of commercial fish species for decades (e.g. Piet and Rice, 2004). Such data are not routinely available for non-target species. However, Greenstreet (submitted) determined values for a number of life-history traits for every species recorded in the Q1 IBTS. This included estimates of the length-at-maturity. Thus it is possible to estimate the proportion of the population biomass of each species exceeding their length-at-maturity (though it should be noted that this was fixed over time for each species, and sexual dimorphism in the size at maturity was not considered), thus providing an analogue to the spawning stock biomass indicator for every species sampled by the survey. We derive this indicator here and examine its potential to fulfil the population demographic indicator role (Table 2.1; indicator 1.3.1).

Two major developments greatly increased the capacity of North Sea fishing fleets to catch fish: the replacement of sail power by steam driven vessels at the start of the 20th century and the subsequent switch to diesel engines in the 1950s. Fish landings increased steadily by a factor of three through the first half of the century before subsequently declining to a level in 2008 lower than at any time in the previous 120 years. Variation in landings per unit fishing power, a better indicator of actual abundance of fish in the sea, was even more profound. This indicator showed a 66% reduction between 1890 and 1920, followed by a recovery that regained perhaps half of this loss by the late 1950s. Between 1955 and 1980 landings per unit fishing power declined by more than 90%, stabilizing from 1980 onwards (Thurstan *et al.*, 2010). Comparison of heavily fished with lightly fished areas within the North Sea suggested that fishing had affected 11 of 12 univariate community metrics applied to the demersal fish assemblage by the middle of the 20th century (Greenstreet and Rogers, 2006). Species with "slow-type" life-history traits (large-bodied, slow growing, late age and large size at first maturity, low fecundity, etc.) are particularly sensitive to additional fishing mortality (Jennings *et al.*, 1998). Populations of many elasmobranch species, a group of species particularly characterised by "slow-type" traits, had declined markedly in the North Sea (Greenstreet and Hall, 1996; Greenstreet *et al.*, 1999; Walker and Hislop 1998; van Strien *et al.*, 2009) and life-history trait composition among the demersal assemblage as a whole had become "faster" (Jennings *et al.*, 1999; Greenstreet & Rogers, 2000) by the 1960s.

Consistent standardised sampling by all participants in the Q1 IBTS, the longest-running groundfish survey still currently operating in the North Sea, only started in 1983. Major changes in the demersal fish assemblage had therefore already taken place well before this survey commenced, so no indication of the state of the assemblage prior to these changes happening can be obtained directly from available survey data. In situations where available data are insufficient to allow indicator target levels to be set empirically (Link, 2005), the setting of trend directions has been proposed as a pragmatic alternative (Jennings and Dulvy, 2005; Shin *et al.*, 2005). The rationale here is that where we have good evidence to suggest that the current state of an ecosystem component is unsatisfactory then we can state with some assurance that an improvement is required, even if we cannot define what a satisfactory state might look like.

Greenstreet (submitted) ranked 119 demersal fish species sampled by the Q1 IBTS by their "sensitivity" scores, derived from an analysis of their life-history traits (L_{inf}, k, L_{mat}, A_{mat}). Low-ranked species with the "slowest-type" traits were considered the most "sensitive" and high-ranked species with the "fastest-type" traits were deemed the most "resilient". The 40 species in the lower 33 percentile were classed as "sensitive" species. For these species trend-based targets can be set for each species-level indicator (Table 2.2). For each indicator we anticipate that mitigatory management should instigate a positive trend.

Greenstreet (submitted) used factor analysis to examine abundance trends in 73 species recorded in at least nine years of the Q1 IBTS but thought likely to have been present in the Greater North Sea throughout the entire 26 years duration of the survey. In both analyses, scores of the first factor, which explained the majority of the total variance, showed monotonic trends. However, only 39 (53%) species loaded onto these factors. Thus for nearly half the species where the data available might be considered sufficient to treat them as potential species-specific indicators, underlying trends in their abundance were not linear. This tends to invalidate the use of standard parametric trend analysis techniques. However an alternative non-parametric approach can be used. The entire duration of each species-specific metric data time-series can be treated as the "reference period". The last year in the time-series can then be considered the "current assessment year". A target position relative to the "reference period" can then be set for the "current assessment year". For example if the metric data for the entire "reference period" are ranked in order of increasing value, then a target could be set such as 'The "current assessment year" metric value should be in the top 15 (or 25, or 35, or 50) percentile of all values in the full time-series "reference period"'.

Figure 2.1 illustrates this by showing temporal variation in the species-specific population abundance indicator of three "sensitive" species. For each species annual metric values falling in the top 15, 25, 35, and 50 percentiles of all data in the full "reference period" time series area shown. Halibut *Hippoglossus hippoglossus* displayed a generally monotonic increasing trend and as a result 3 or 5 of the latest 7 data points in the time-series lie in the top 15 and top 25 percentiles respectively of the full time-series "reference period". This species would have met regularly its species-specific metric target in recent years. Anglerfish *Lophius piscatorius* showed a quite different trend, increasing in abundance in the first half of the time-series and decreasing again in the second part. Anglerfish is clearly not meeting its species-specific metric target in recent years and, on the basis of this trend, shows little inclination to do so. Spotted ray *Raja montagui* displayed yet a third type of trend, varying in abundance quite markedly over most of the time-series with little indication of any obvious trend. At the end of the time-series, however, several relatively high values occur and in the last "current assessment year" this species-specific metric would meet a target set at the top 15 percentile. However, spotted ray illustrates potential shortcomings of the method, if a species has fundamentally no time trend but strong inter-annual variation. A species of this type might be expected to meet or fail the target in subsequent years purely by "chance". This issue is dealt with below.

EC (2010) clearly states that whether or not good environment status is achieved is determined on the basis of the criteria of each Descriptor (Table 2.1). To meet good environmental status may require all indicators linked to each criterion to meet their target, or just a specified number of indicators. However, at this point we have only considered targets for the individual species-specific metrics. How many of these metrics need to meet their defined upper percentile targets in order to conclude that objectives for the species-level indicator had successfully been achieved? A similar situation existed for the North Sea EcoQO for commercial fish species, which states "Spawning stock biomass of commercial fish species in the North Sea should be above precautionary reference points for commercial fish species where these have been agreed by the competent authority for fishery management" (Heslenfeld and Enserink, 2008). At least 17 stocks met this definition, but what number of stocks should meet this target for the overall EcoQO to be achieved? Analysing the performance of these indicators, Piet and Rice (2004) concluded that a false alarm rate approaching 25% would be likely in the current fisheries regime. Deciding that 100% of stocks should exceed their individual precautionary biomass levels therefore involves a high risk of apparent failure because of false alarms, even if all stocks were indeed above their precautionary biomass level. Setting a more risk-averse target of 80% of stocks should meet their specific targets might be a politically more acceptable EcoQO. So, of the 40 potential sensitive species-specific metrics for each indicator, what number (proportion) need to meet their stated upper percentile targets to conclude that the indicator objective had successfully been met?

If the probability of a particular event happening can be determined, the binomial distribution allows us to predict the number of such events that should occur in a given number of trials. Under the stationarity and independence assumption, the probability of any particular datum falling in the top 15, 25, 35, of 50 percentile of any set of data might be anticipated to be ≤15%, ≤25%, ≤35% and ≤50% respectively. But in this case we are dealing with time-series data, so that successive values are auto-correlated to some unknown degree. The probability of the last datum in a time-series falling in any specified upper percentile range (assuming no drift over time took place) was therefore estimated using a Brownian random walk model with 20 000 time steps, incrementing the simulated metric value by a normally distributed random number at each time step. The process was repeated to 200 000 times for each target percentile range to estimate the corresponding probabilities for the 26 years time series (Table 2. 3). Now that the probability of observing a "current assessment year" metric value falling in any specified upper percentile target range under the null hypothesis of a random step progression is known, the binomial distribution can be used to ascertain how many such events should be observed for any particular number of species-specific metrics for there to be a less than 5% probability of this happening by chance. These numbers represent the indicator-level target. If this, or higher number, of metric targets are met, then the indicator target can be considered to have been successfully achieved as there is an underlying time trend.

An important point to note here is that, because the individual species-specific metrics targets are trend-based, achieving particular indicator-level targets would not necessarily mean that GES had been achieved in respect of that indicator. We still do not know what the GES state, as measured by that indicator, looks like. What such a result does tell us is that the number of sensitive demersal fish species now on recovery trajectories is significantly more than the number that might be expected simply by chance if no change happened. This could be interpreted as implying that management measures implemented to mitigate human impacts on fish stocks are successfully moving the selected "sensitive" species component of the assemblage the right direction.

A second point to understand is that the upper percentile range target level for the species-specific metrics and the overall significant departure from the binomial distribution indicator-level target are balanced. Setting more stringent metric-level targets, simply means that fewer "sensitive" species have to meet this target in order for any departure from the binomial distribution to be statistically significant. In many ways this behaviour is advantageous as exploration of this balance point may allow the most risk-averse (the risk of false alarms, i.e. Piet and Rice, (2004)) condition to be determined. If, on the other hand, this is deemed undesirable – politically one might wish to set quite stringent target levels for the individual species-specific metrics, yet still retain a more ambitious overall indicator-level target – then the significance level for the binomial distribution departure can simply be adjusted: from the 5% probability level to perhaps the 1% probability level (taking into account how many species are considered).

This process, using the six species-specific metrics listed in Table 2.2, was applied to the Q1 IBTS data set to assess recent trends in the North Sea demersal fish assemblage. Firstly, although Greenstreet (submitted) classified 40 species as "sensitive", data were only considered adequate for analysis in respect of 27 of them. The criterion on which this decision was based was that a species should be recorded in at least half of the years of the time-series to be included (i.e. 13 years in the present case study). Each year from 2001 to 2008 was in turn considered the "current assessment year", and in each case the "reference period", used to define the upper percentile range targets, was the full time series from 1983 up to and including the year in question.

Figure 2.2 shows for each MSFD indicator class the number of individual species-specific metrics meeting particular upper percentile targets and Figure 2.3 shows the binomial probability of observing this number, or higher, with a probability of

P<0.05. Four indicator classes, distributional range, population abundance, population biomass and population demographic, showed relatively consistent trends with variable but low numbers of species-specific metrics meeting their targets over the period 2001–2006, followed by a marked increase in 2007 and 2008. Despite this increase, the number of species-specific metric targets being met still remained lower than the number required to demonstrate a significant departure from the binomial distribution at the 5% significance level. So the indicator-level target has yet to be met, but the trend was clearly moving in the right direction in the two most recent years. The two species-specific metrics supporting the distribution pattern within range indicator both showed highly variable behaviour, with the number of metrics meeting their upper percentile range targets being sufficiently high in a number of years as to achieve statistical significance with respect to the indicator-level target. However, no trend was apparent and there was no consistency as to whether or not the indicator target was met. This result suggests that performance of the mean:variance metric may not be sufficiently well understood and further developmental work may be required to make these particular metrics operational. Alternative metrics could also be tried, such as the proportion of the range occupied by a specified proportion of the population (e.g. Blanchard *et al.*, 2005).

The approach above provides for an evidence-based method for setting targets for the more routinely sampled demersal fish that may be considered "sensitive", although there are still several issues that require further consideration, including the appropriate selection of variance in the random walk (and the autocorrelation if it is not 1).

Table 2.2. Derivation of trends-based targets for each species-specific metric applied to "sensitive" demersal fish species in the North Sea.

EC Indicator	Metric	Perturbed state condition	Trend target
1.1.1 Distributional range	Proportion of surveyed ICES rectangles occupied	Depressed – population reduced and only prime habitat sites occupied	+ve
1.2.1 Distributional pattern within range	Mean:variance ratio of abundance in occupied rectangles by numbers	Depressed – Density remains high in prime habitat site, but few occupied marginal habitat sites hold low densities leading to high variance	+ve
1.2.1 Distributional pattern within range	Mean:variance ratio of abundance in occupied rectangles by biomass	Depressed – Density remains high in prime habitat site, but few occupied marginal habitat sites hold low densities leading to high variance	+ve
1.2.1 Population abundance and/or biomass	Abundance	Depressed – population abundance reduced by unsustainable mortality	+ve
1.2.1 Population abundance and/or biomass	Biomass	Depressed – population biomass reduced by unsustainable mortality	+ve
1.3.1 Population demographic characteristics	Proportion of biomass greater than length-at-first maturity	Depressed – spawning component in slow growing late maturing species reduced by unsustainable mortality	+ve

Table 2.3. For a given number of species-specific metrics, the number (and percentage in parenthesis) of metrics required to meet specified upper percentile targets in order for the probability of observing such a departure from the binomial distribution being less than 5%.

Number of species-specific metrics	Metric values in "current assessment year" should lie within the upper:			
	15% of all values (*P* = 0.253)	25% of all values (*P* = 0.332)	35% of all values (*P* = 0.403)	50% of all values (*P* = 0.500)
10	6 (60%)	7 (70%)	8 (80%)	9 (90%)
15	8 (53%)	9 (60%)	10 (67%)	12 (80%)
25	11 (44%)	13 (52%)	15 (60%)	18 (72%)
35	14 (40%)	17 (49%)	20 (57%)	23 (66%)
40	16 (40%)	19 (48%)	22 (55%)	26 (65%)
50	19 (38%)	23 (46%)	27 (54%)	32 (64%)

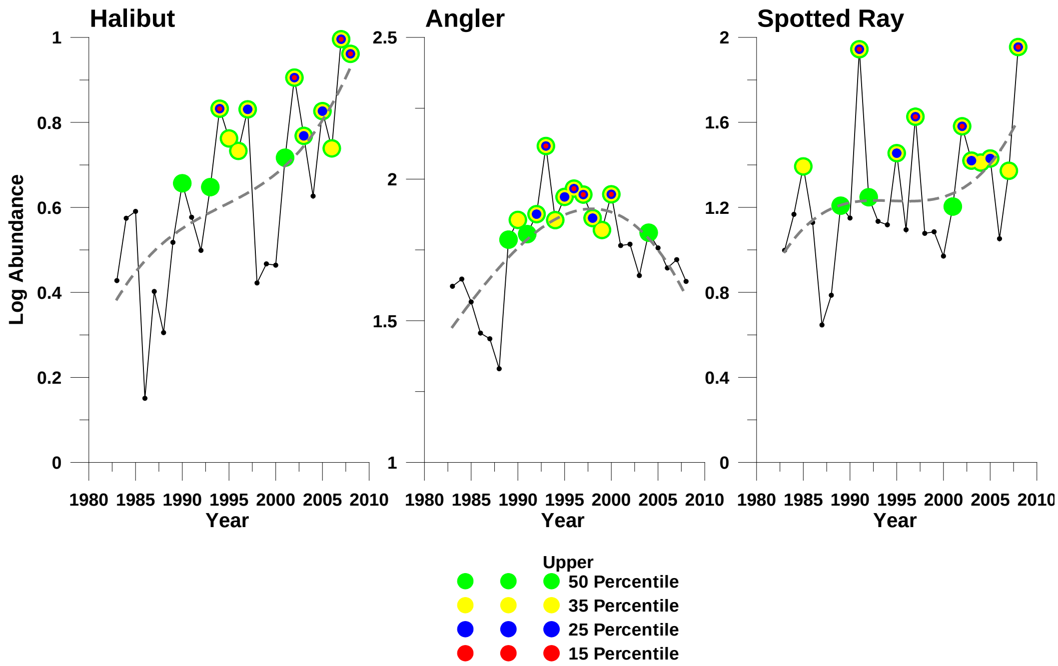

Figure 2.1. Variation in values of the species-specific population abundance indicators for three sensitive species with different temporal trends. Annual indicator values falling in the upper 15, 25, 35, and 50 percentiles of all values in the whole time-series are indicated. Grey dashed line is a 3rd degree polynomial fitted to the data to highlight underlying trends.

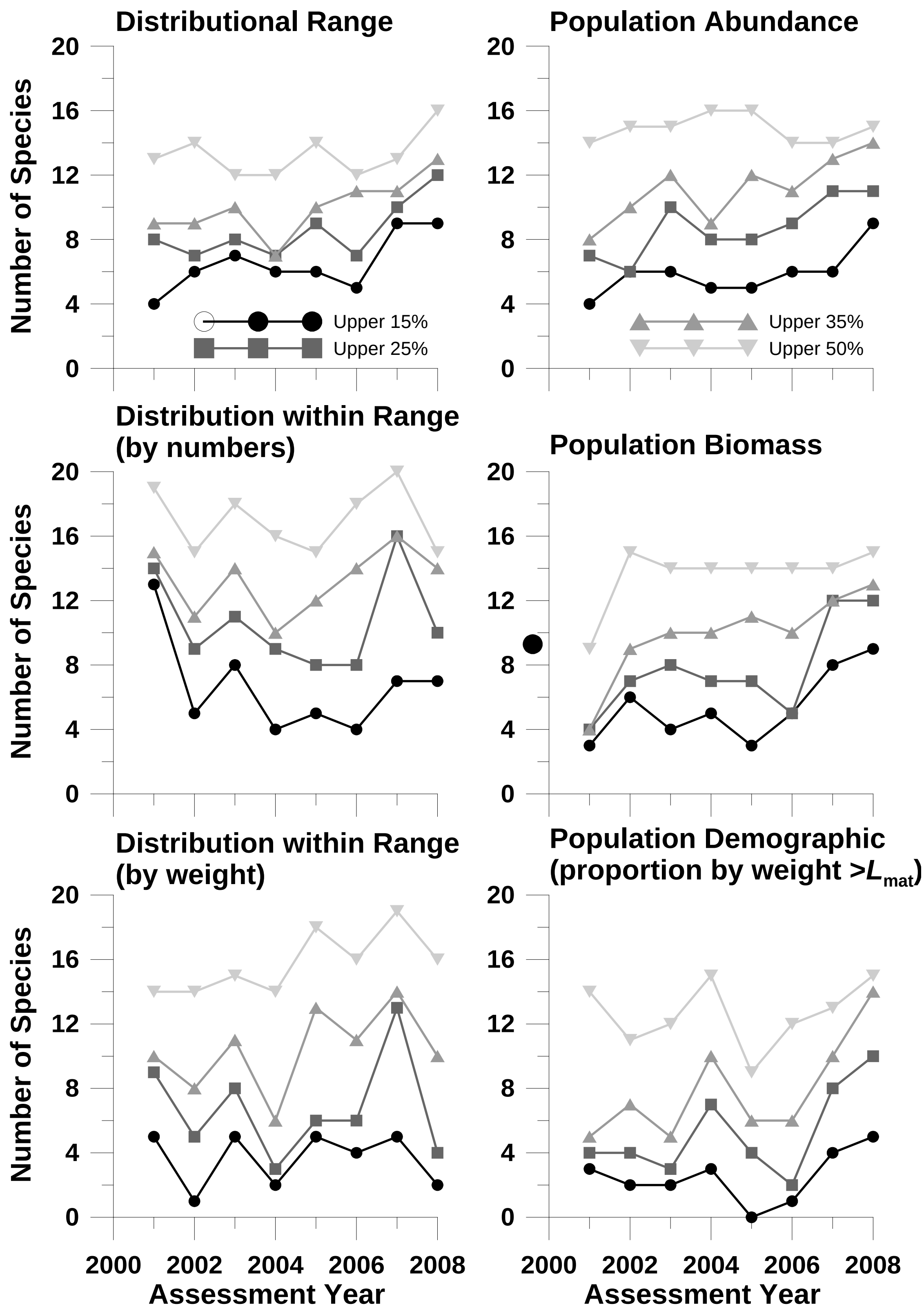

Figure 2.2. Number of individual species-specific metrics meeting specified metric targets (The "current assessment year" metric value should be in the top 15 (or 25, or 35, or 50) percentile of all values in the full time-series "reference period") for each species-level indicator class evaluated.

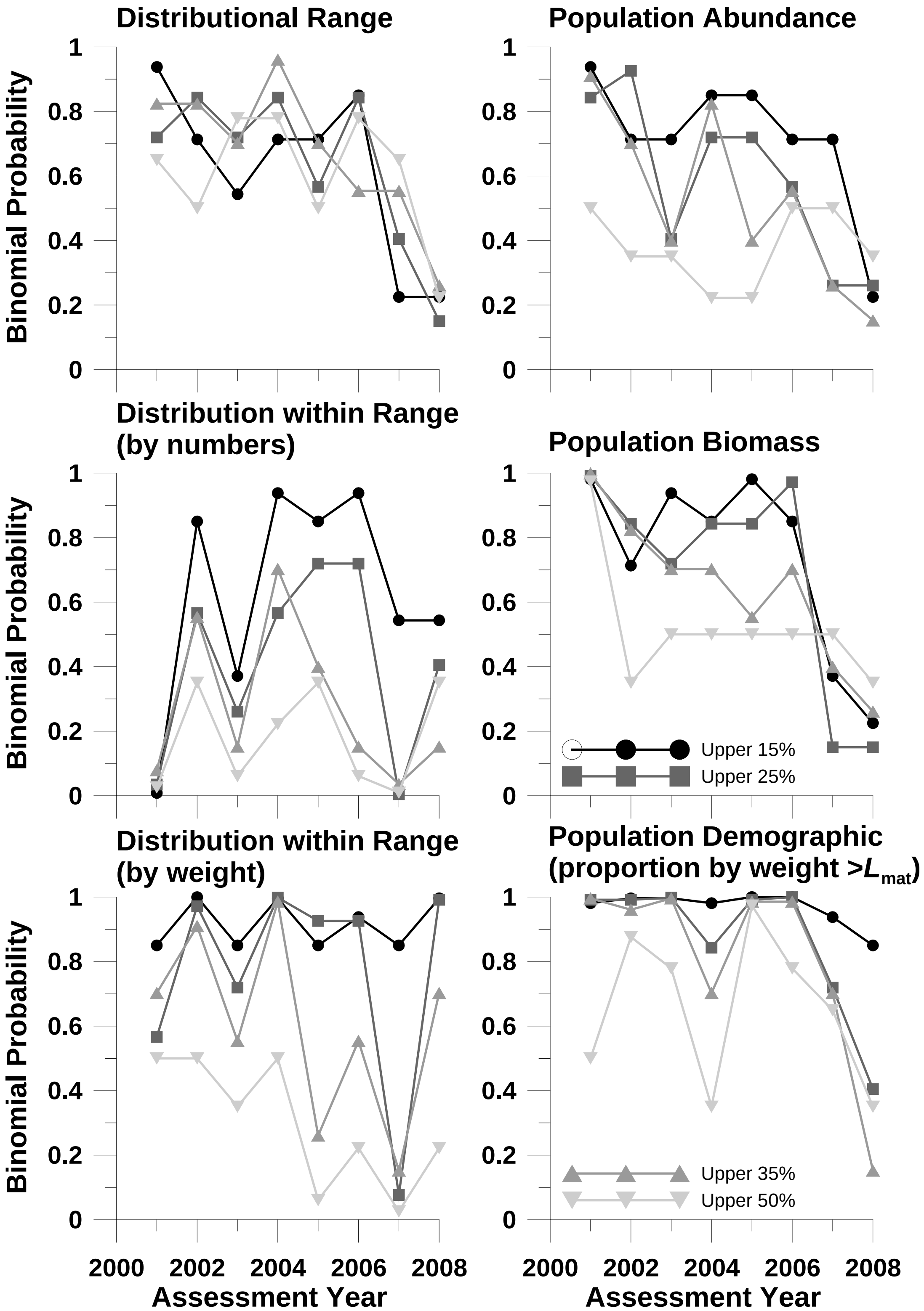

Figure 2.3. Probabilities associated with the observed number of species-specific metrics meeting specified percentile range targets derived from the binomial distribution, knowing the probability that any one metric will meet its target (Table 2.2) and given 27 species-based metrics.

2.4.4 Case study: Review of indicators estimating the distributional range and patchiness of species (or species groups) in the Bay of Biscay and the Barents Sea

Five indicators in the descriptor 1 of the MSFD refer to the spatial distribution (range, pattern, and extent) of biodiversity features. Two of these indicators are based on species or groups of species (D1.1.1 and D1.1.2) whereas the three others refer to habitats (D1.4.1, D1.4.2, and D1.5.1). The present case study briefly reviews some spatial metrics that could be used to inform two of the five indicators, and test the relevance of these metrics using three datasets displaying different sampling strategies and ecological properties: French bottom trawl scientific surveys conducted in the Bay of Biscay, French fisheries onboard observation program in the Bay of Biscay, and Joint Norwegian-Russian ecosystem scientific surveys conducted in the Barents Sea (Anon 2011).

Quantification of the link between ecological patterns and ecological functions and processes is the main aim of the field of spatial ecology. The increasing popularity of that ecological field is mainly due to the awareness of the importance of considering spatial patterns in explaining species distribution and the alteration of the environment which requires the understanding of its heterogeneity (Fortin and Dale 2005). There are thus multiple approaches developed to assess and study the spatial patterns in the literature (see review of Fortin and Dale 2005; and also Legendre & Fortin 1989; Wackernagel 1998; Chiles & Delfiner 1999). Among these approaches, two main fields of research have been extensively applied in terrestrial ecology (landscape ecology) and to fisheries and fish ecology (geostatistics).

Historically rooted in terrestrial ecosystems, landscape ecology has developed multiple tools and methods to quantify spatial heterogeneity. Spatial pattern metrics have proved to be very useful in monitoring environmental changes and for studying the multi-scale processes that drive organism distributions and biodiversity (Botequilha Leitão *et al.* 2006, Mizerek *et al.* 2011). However, the majority of this literature has been developed for the analyses of terrestrial habitat maps, yet the relevance of such metrics in the marine environment is poorly known.

Wedding *et al.* (2011) recently reviewed the number of studies using landscape ecology spatial metrics in the marine environment. They found that over the past 30 years a total of 28 studies quantified spatial patterns using these spatial metrics. Most of the studies were conducted on habitats of sessile marine organisms (coral reefs, seagrasses, mangroves, coastal wetlands) easily sampled using satellite images (Manson *et al.* 2005, Pittman *et al.* 2004, Drew & Eggleston 2008, Meynecke *et al.* 2008). A large proportion of these studies also used the spatial metrics to test the effect of the seascape configuration on the spatial distribution of organisms. Yet, the use of these metrics to quantify the changes in marine species distribution is still quite new and untrialled.

<u>Type of data and spatial metrics</u>: Spatial metrics can be categorised according to the types of data required to compute them: patch-based and point-based data (Gustafson 1998). The patch-based data, also called categorical maps, is composed of maps of categorical variables (e.g. seagrasses, species abundance categories) identifying patches that are spatially delineated and relatively homogeneous with respect to the selected scale of the study. The second type of data, the point-based data, is composed of a collection of sample points randomly or systematically distributed in space. Metrics computed on that type of data assume that the variable under study is spatially continuous whereas metrics calculated on the patch-based data assume that the variable exhibits abrupt transitions (boundary) to adjacent areas (patches).

Data from scientific surveys usually come in the form of a yearly random point-based dataset. However, as mentioned earlier the spatial metrics were initially developed for patch-based data that is to be calculated on categorical maps. Therefore, survey data must be transformed in categorical maps prior to the computation of the patch-based metrics. As the present work is a preliminary study aiming at testing the metrics, we chose to develop species maps using constrained cluster analyses for untransformed count data.

Scaling factors; grain, extent and thematic resolution: The spatial metrics will be affected by three scaling factors: the grain, the extent and the thematic resolution. The grain is the resolution of the data (i.e. the smallest distance between two sampling stations), the extent refers to the size of the studied area, and the thematic resolution is the amount of details in a map as defined by the number of classes (or groups). As Turner (1989) and Wedding *et al.* (2011) thoroughly underlined in their papers, several spatial metrics are sensitive to all or some of the three scaling factors. For instance, increasing the grain of a study without modifying its extent will result in decreasing several spatial metrics such as the number of patches (NP). Therefore, great care should be taken in choosing 1) the right spatial scales (grain and extent) prior to the metric's computation and 2) a relevant method to identify the number of classes included in a categorical map.

All of the patch-based landscape metrics reviewed in this study come from Gustafson (1998) and McGarigal and Marks (1995), whereas point-based metrics are new metrics suggested for this study to characterize landscapes. We categorized the patch-based metrics in two classes: those describing the spatial composition and those estimating the spatial configuration of the patches or the landscape. Spatial composition stands for the patch type and abundance while spatial configuration refers to the patch distribution, orientation and isolation. A total of eighteen metrics were chosen and computed on the three case studies (Table 2.4). As these metrics represent only a subset of all existing spatial metrics, we invite the reader to read McGarigal and Marks (1995) for a thorough list of landscape metrics. Several descriptions of the chosen metrics come from these authors.

Datasets

French bottom-trawl scientific survey: IFREMER carries out each year since 1987 the EVHOE bottom trawl survey on the eastern continental shelf of the Bay of Biscay. The survey area ranges between 48°30'N in the north and the northern margin of Gouf de Cap Breton in the south (43°N). A 36/47 GOV trawl is used with a 20 mm mesh codend liner. The haul duration is 30 minutes long with a towing speed of 4 knots. Fishing is restricted to daylight hours. Catch weights and catch numbers are recorded for all species. However, for the purpose of this study three species displaying different temporal trends were chosen: argentine *Argentina sphyraena* displaying a general decreasing trend over the years, red gurnard *Aspitrigla cuculus* displaying stable abundances over the years but decreasing trends in body size, and hake *Merluccius merluccius* for which increasing abundances have been observed since 2000.

The EVHOE survey uses a randomly stratified sampling protocol. The sample points are therefore geographically different each year implying that the sampling grain of the study is not constant. In order to compare the indicator's variability over time without being influenced by the effect of the changing grain over time, we chose the years for which we had sufficient common points to do the analyses. The search of common sampling points among years was done using a grid of 0.3° by 0.3° and selecting the points within each cell. Therefore, the indicators of the French bottom-trawl surveys in the Bay of Biscay were computed on the data from 1989, 1990–1992, 1994–1995, 1997, and 2009 on a dataset of 26 common sampling points ranging from 43°00'N to 48°00'N and 7°00'W to 1°00'W. The geographic coordinates were transformed in decimal degrees prior to the computation of the metrics.

French fisheries onboard observation program: The French at-sea observation program started in 2002 for the *Nephrops* fisheries and was extended to all the fishing fleets in 2003. It allows the collection of detailed, geographically co-ordinated information on the fishing effort (haul duration), catches and discard at sea as well as time of year and time of day. The observers collect the fisheries-related data on board French vessels. The data are compiled by fishing operation (i.e. for each gear-shot event). Each fishing operation is characterized by a boat, a gear targeting one or a group of species at a specified time and in a specific geographical location. For the present study, we extracted the data for the three species previously selected from the EVHOE survey (*Argentina sphyraena, Aspitrigla cuculus, Merluccius merluccius*).

The geographic positions of fishery operations from the onboard observation dataset are obviously changing every years. A search for common fishery locations between the years was conducted using the same methodology as the one used in EVHOE, i.e. by creating a grid of 0.11° by 0.11°. The dataset from the onboard observation used in this study was thus composed of 72 points ranging from 43°00'N to 48°00'N and 7°00'W to 1°00'W for the period 2007–2010. The geographic coordinates were transformed in decimal degrees prior to the computation of the metrics.

Joint Norwegian-Russian Barents Sea ecosystem scientific surveys: The indicators were computed on a data set collected during the Joint Norwegian-Russian ecosystem survey during August/September 2004–2009. The survey area covers the whole Barents Sea shelf including the northern part dominated by arctic fishes. A Campelen demersal trawl is used. The haul duration is 15 minutes long with a towing speed of 3 knots. Fishing is done both day and night. Catch weights and catch numbers are recorded for all species. For the purpose of this study four of the most common arctic species were selected since they are likely to be sensitive to increasing water temperatures. The study years (2004–2009) have been the warmest on record (since 1900) and the temperature peaked in 2007. A decline in temperature is expected for the next few years as part of inter-annual fluctuations on an increasing trend. The average distribution of the selected species along with a description of their basic biology is modified from the Atlas of the Barents Sea fishes (Wieneroither *et al.*, 2011). However, it should be noted that arctic fishes are poorly studied (see Mecklenburg *et al.*, 2002, 2011 for reviews on Arctic fishes).

Data analyses: The three datasets were initially composed of a collection of sampling points regularly distributed in space (i.e. fixed grain among years) in which different variables were sampled (e.g. fish abundance, or biomass). In order to compute all the metrics (patch and point-based), it was essential to transform the surveys point-based dataset into patch-based dataset. Figure 2.4 displays the steps used to do that transformation. As mentioned above, there exist several techniques aiming to develop species map distribution based on sampling points (e.g. constrained clustering, geostatistical analyses, home range analyses, habitat mapping) and each of them have different assumptions. As the present work is a preliminary study, we chose to develop species map using constrained cluster analyses. In the constrained clustering method used in here, the analysis is conducted in two stages. At the first stage, the dataset is classified into different groups using regular clustering techniques; then spatial adjacency is examined at the second stage, where those adjacent members that are in the same group are combined into patches (Legendre *et al.* 2002). Although we did not have the time in here to compare different techniques to develop species distribution maps, such comparisons should imperatively be done to test the sensitivity of the metrics in regard to the chosen mapping method.

All the metrics were computed using R (R Development Core Team 2010). It is worth mentioning that all the patch-based metrics are included in FRAGSTAT (McGarigal and Marks 1995), a free-distributed computer software program designed to compute a wide variety of landscape metrics for categorical map patterns (McGarigal and Marks, 1995). Several of these landscape metrics were also recently implemented in the R package SDMTools v1.1–9. Point-data metrics were computed using the sp package v0.9–95 (Pebesma & Bivand 2012).

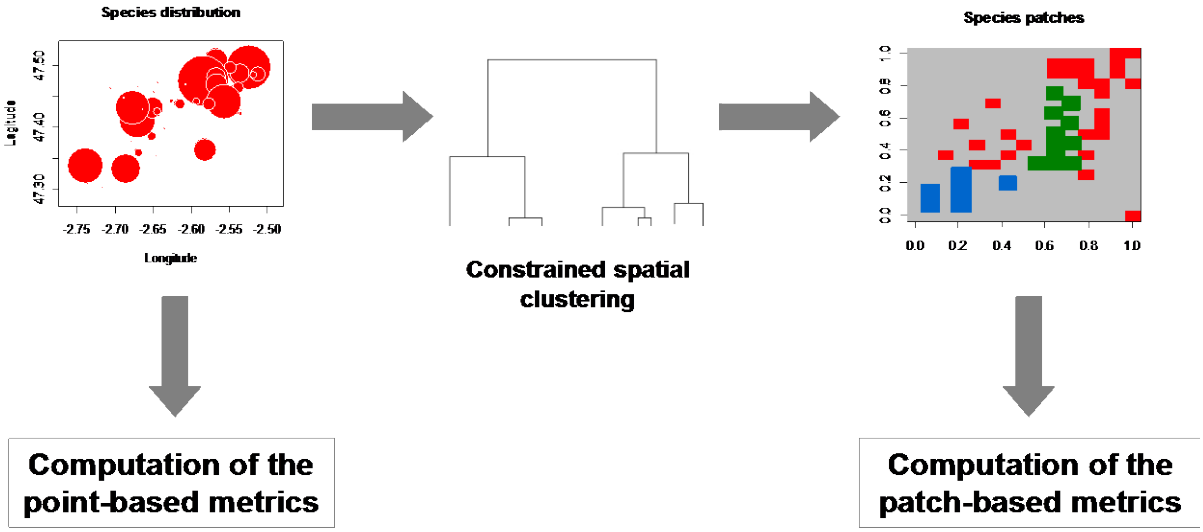

Figure 2.4. Flowchart presenting the steps required for the transformation of point-based data into patch-based data. The constrained spatial clustering was used for the purpose of this study. However, several other techniques modelling species distribution are available and should be tried.

<u>Bottom-trawl survey and onboard observation program in the Bay of Biscay</u>: The spatial distributions of the three species observed through the filter of the 26 common sampling points over the selected time period for the EVHOE and through the 72 points for the onboard observations datasets are shown in Figures 2.5 and 2.6, respectively. A general decrease in spatial distribution of argentine in the 1990s can be observed but the distribution in 2009 clearly contrasts with the other years (Figure 2.5). The spatial distribution of red gurnard was highly variable between years, reaching higher density values in the middle and northern parts of the bay. Similar northern distributions were displayed by hake, although that species showed a more coastal distribution in comparison to the gurnard.

The spatial distribution of the same three species for the onboard observations dataset cannot be directly compared to the EVHOE's species distributions, as the selected years in the two time series do not perfectly overlap. Figure 2.6 shows that the onboard observations sampling effort is mostly concentrated in two patches; one patch located in the southern Bay of Biscay, near the Gironde estuary. and the other one in the northern bay on the Grande Vasière (a *Nephrops* ground). Very little information regarding the distribution of argentine and red gurnard is given by these maps. Hake displayed a similar distribution to the one in EVHOE, despite the fact that the two time series covered different years. This is however not surprising as the Grande Vasière is a well known nursery for hake.

Spatial metrics: Several spatial metrics computed for argentine using the EVHOE bottom-trawl survey displayed decreasing trends, except for the metrics estimating the diversity of patches (SIDI and SIEI). This is partly due to the low values observed in the last part of the time series (after 2000; Figure 2.7). This analysis is however very delicate to interpret as there is only one year included in the analyses after 2000. Nevertheless, these decreasing trends are more pronounced for the metrics assessing the spatial configuration of the patches (e.g. NP, PD, PSSD) and the point-based metrics (D_i, ASS). Indicators computed for red gurnard did not show any obvious trends, rather they displayed high variability (e.g. composition metrics) or high stability (e.g. MPS, PSSD, SHAPE, LATmin and max) over the time series. The hake's spatial metrics were also highly variable in the 1990s but seems to show a slight increase in 2009 for some metrics (e.g. CA, SIDI, PSSD, SHAPE, Di) and a slight decrease for others (PRD and SIEI). However, this result is, as discussed for the argentine, highly sensitive to the low number of years after 2000.

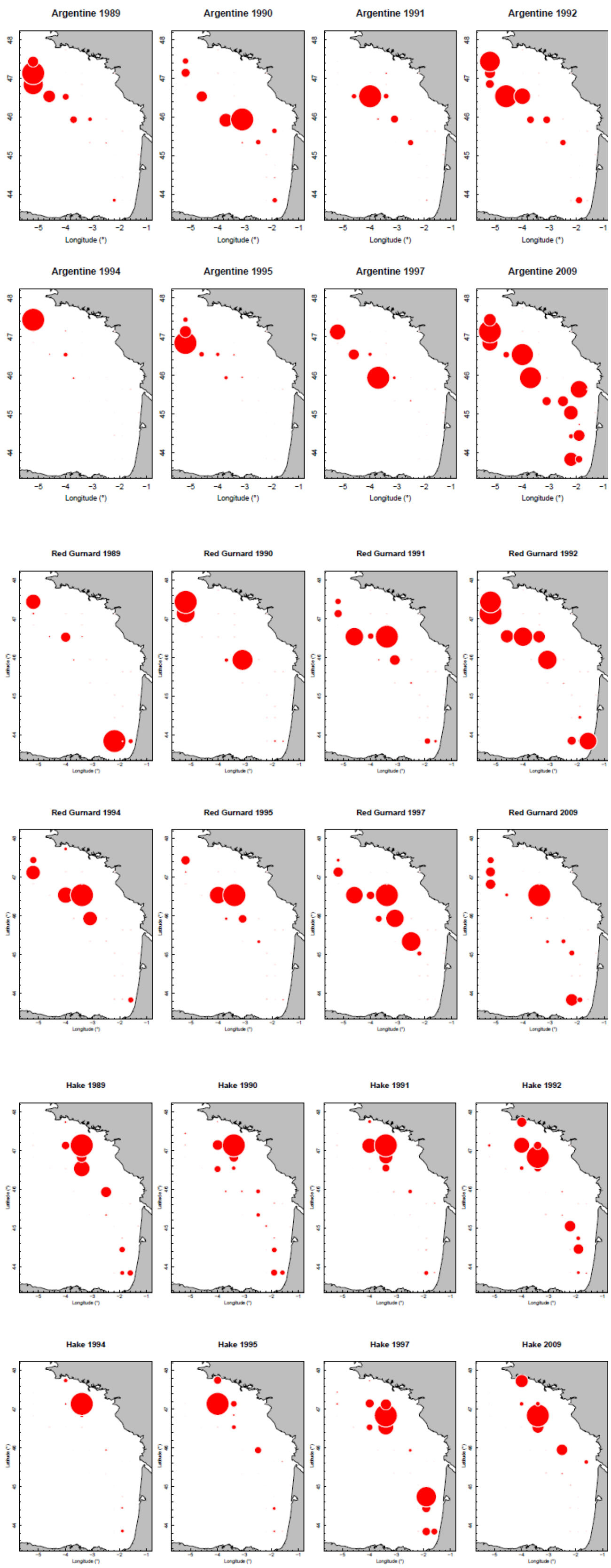

Figure 2.5. Distributions of *Argentina sphyraena* (upper left), *Aspitrigla cuculus* (upper right) and, *Merluccius merluccius* (lower) in the Bay of Biscay between 1988–2009 using the EVHOE dataset. The size of the red bullets is proportional to the average number of individuals caught per km towed.

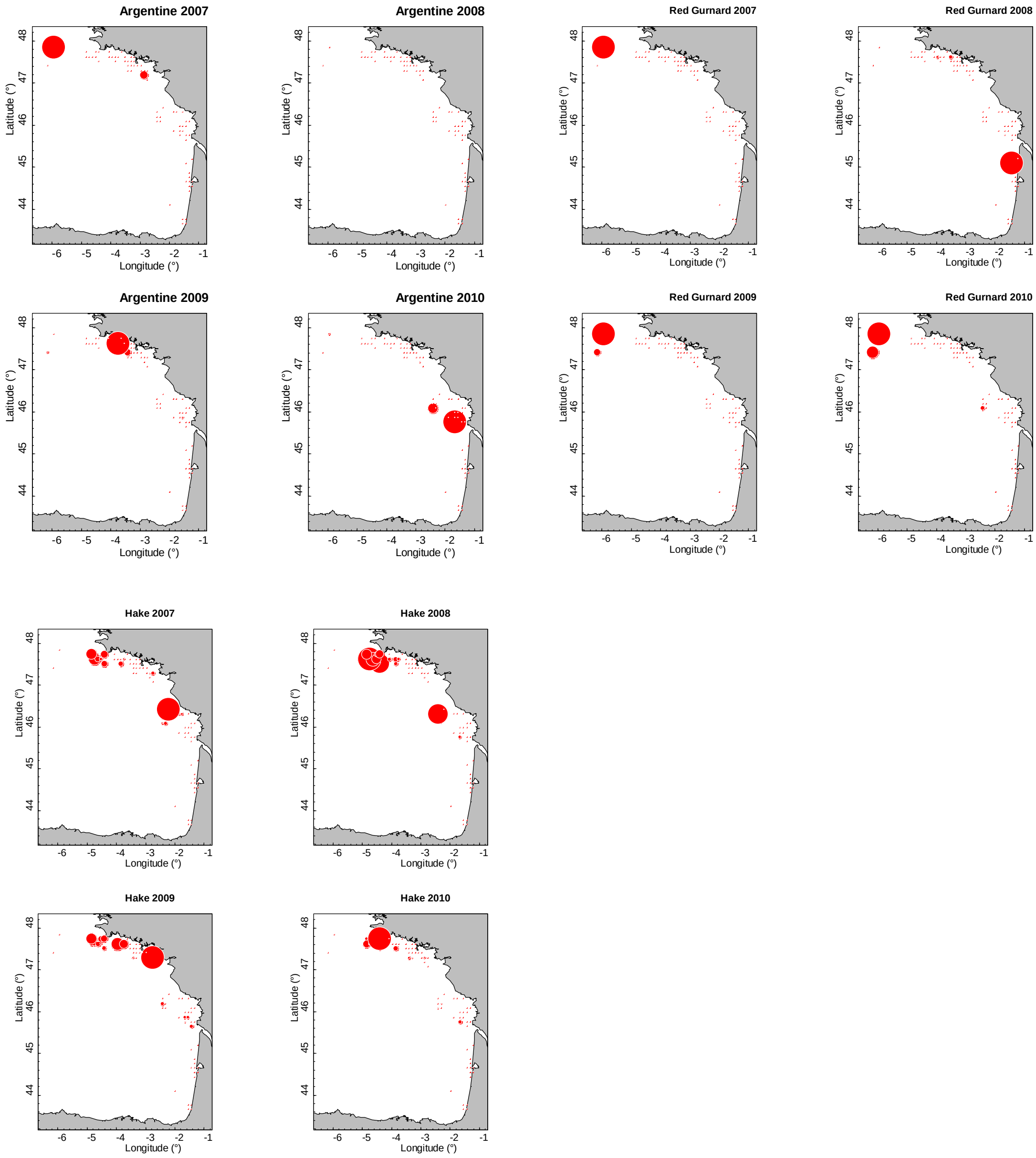

Figure 2.6. Distributions of *Argentina sphyraena* (top left), *Aspitrigla cuculus* (top right) and *Merlucius merlucius* (bottom left) in the Bay of Biscay between 2007–2010 using the dataset from the onboard observation program (OBSMER). The size of the red bullets is proportional to the average number of individuals caught per km towed.

Table 2.4. Selection of metrics quantifying different aspects of the spatial structure of a variable of interest. Metrics were calculated using abundance data.

Representation	Type of data	Spatial aspects	Metrics	Description
Categorical maps	Patch-based data (i.e. maps)	Spatial composition	Class area (CA)	Area (m2) of the patch and it is thus limited by the grain and extent of the image
			Percent of landscape (%LAND)	Sum of the areas (m2) of all patches of the corresponding patch type, divided by total landscape area (m2), multiplied by 100 (to convert to a percentage)
			Patch richness (PR)	Number of different patch types present within the landscape boundary
			Patch richness density (PRD)	Number of different patch types present within the landscape boundary (PR) divided by the total area of the landscape (TA)
			Simpson's diversity (SIDI)	SIDI equals 1 minus the sum, across all patch types, of the proportional abundance of each patch type squared.. The SIDI = 0 when the landscape contains only 1 patch (i.e., no diversity). SIDI approaches 1 as the number of different patch types (i.e., patch richness, PR) increases and the proportional distribution of area among patch types becomes more equitable
			Simpson's evenness (SIEI)	Observed Simpson's Diversity Index (SIDI) divided by the maximum Simpson's Diversity Index for that number of patch types. It is computed as 1 minus the sum, across all patch types, of the proportional abundance of each patch type squared, divided by 1 minus 1 divided by the number of patch types
Categorical maps	Patch-based data (i.e. maps)	Spatial configuration (patch relative to itself)	Number of patches (NP)	Number of patches of the corresponding patch type. The NP equals 1 when the landscape contains only 1 patch of the corresponding patch type; that is, when the class consists of a single patch
			Patch density (PD)	Number of patches of the corresponding patch type (NP) divided by total landscape area (TA)
			Mean patch size (MPS)	Sum of the areas (m2) of all patches of the corresponding patch type, divided by the number of patches of the same type. The range in MPS is limited by the grain and extent of the image and the minimum patch size in the same manner as patch area
			Patch size standard deviation (PSSD)	Square root of the sum of the squared deviations of each patch area (m 2) from the mean patch size of the corresponding patch type, divided by the number of patches of the same type. Note that this is the population

				standard deviation, not the sample standard deviation
			Patch shape complexity (SHAPE)	Patch perimeter (m) divided by the square root of patch area (m2), adjusted by a constant to adjust for a circular standard (vector) or square standard (raster). Therefore, the SHAPE = 1 when the patch is circular (vector) or square (raster) and increases without limit as patch shape becomes more irregular
		Spatial configuration (patch relative to the neighbourhood)	Patch cohesion (COHESION)	COHESION equals 1 minus the sum of patch perimeter (in terms of number of cell surfaces) divided by the sum of patch perimeter times the square root of patch area (in terms of number of cells) for patches of the corresponding patch type, divided by 1 minus 1 over the square root of the total number of cells in the landscape, multiplied by 100 to convert to a percentage. Note, total landscape area (TA) excludes any internal background present. Patch cohesion increases as the patch type becomes more clumped or aggregated in its distribution; hence, more physically connected
Point-data analyses	Collection of samples (point-based data)	Continuous spatial structure	Latitudinal range (LAT)	Minimum and maximum values of the latitude in which the variable appears in the dataset
			Dispersion Index (Di)	Variance to mean ratio indicating the “clumpiness” of the landscape. Large values (Di >1.0) correspond to existence of "patches" whereas small values (Di < 1.0) correspond to a more-uniform-than-random distribution (i.e. “even” or “uniform” distribution)
			Number of spatial scales (NSS)	Number of significant spatial scales. The later being identified using Moran’s Eigenvector Maps (MEM, (Dray *et al*. 2006)). MEM approach is based on spectral decomposition of spatial relationships among sampled sites (neighbouring) and is related to autocorrelation structures through the Moran’s index. Therefore each scale corresponds to a specific wavelength.
			Average size of the spatial scales (ASSS)	Mean length of the significant spatial scales (km). As the MEM is a spectral decomposition, the length of a specific scale (λi) is approximated using the formula proposed by Guenard *et al.* (2010): $\lambda i = 2 (L + s) / i + 1)$ where L is the extent of the sampled section and s is the sampling interval
			Standard deviation of the spatial scales (SSSD)	Standard deviation of the size of the spatial scales (km)

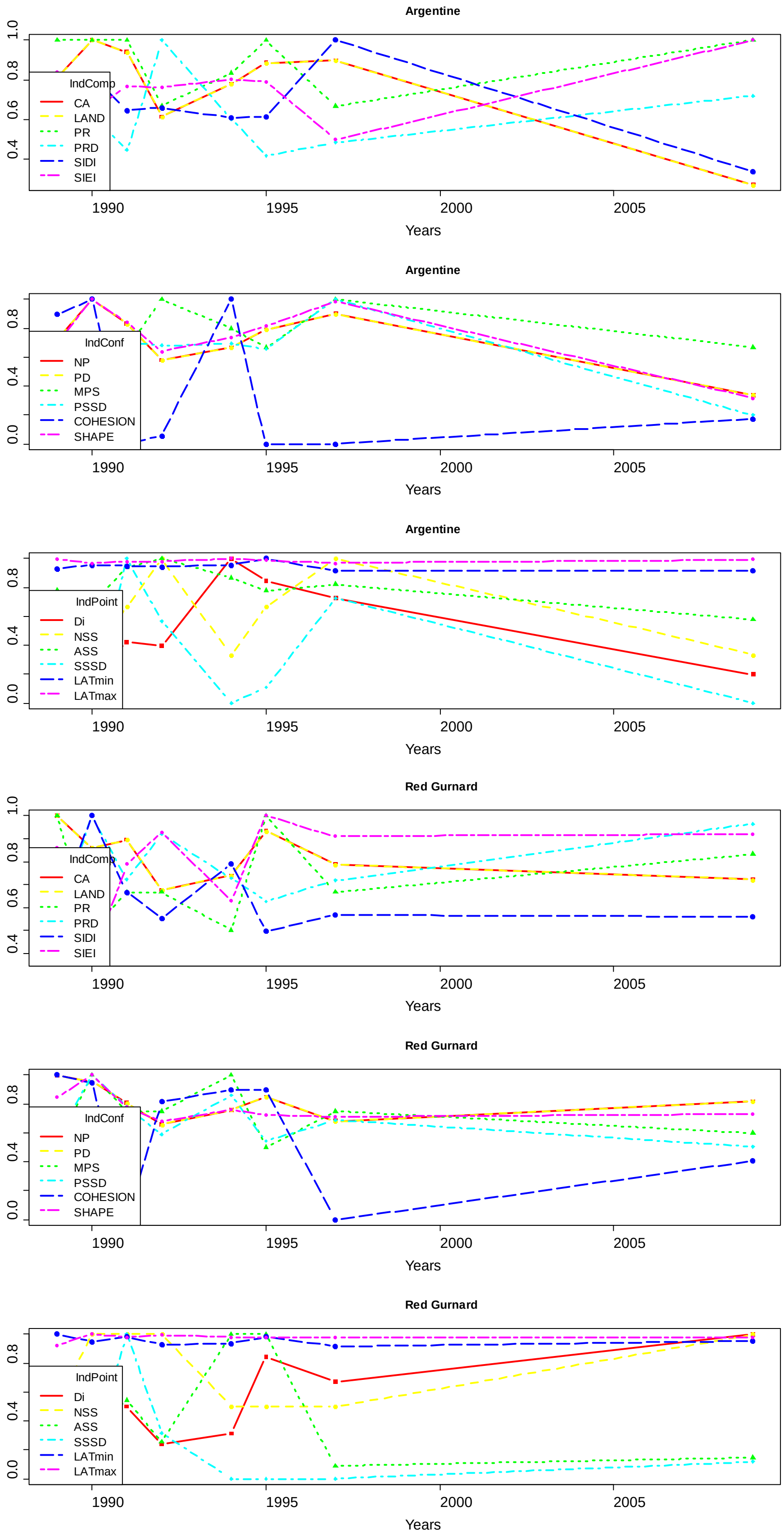

Figure 2.7. Spatial metrics computed for *Argentina sphyraena* (top) and *Aspitrigla cuculus* (bottom) in the Bay of Biscay between 1988–2009 using the EVHOE dataset. No data analysed for 2001–2008.

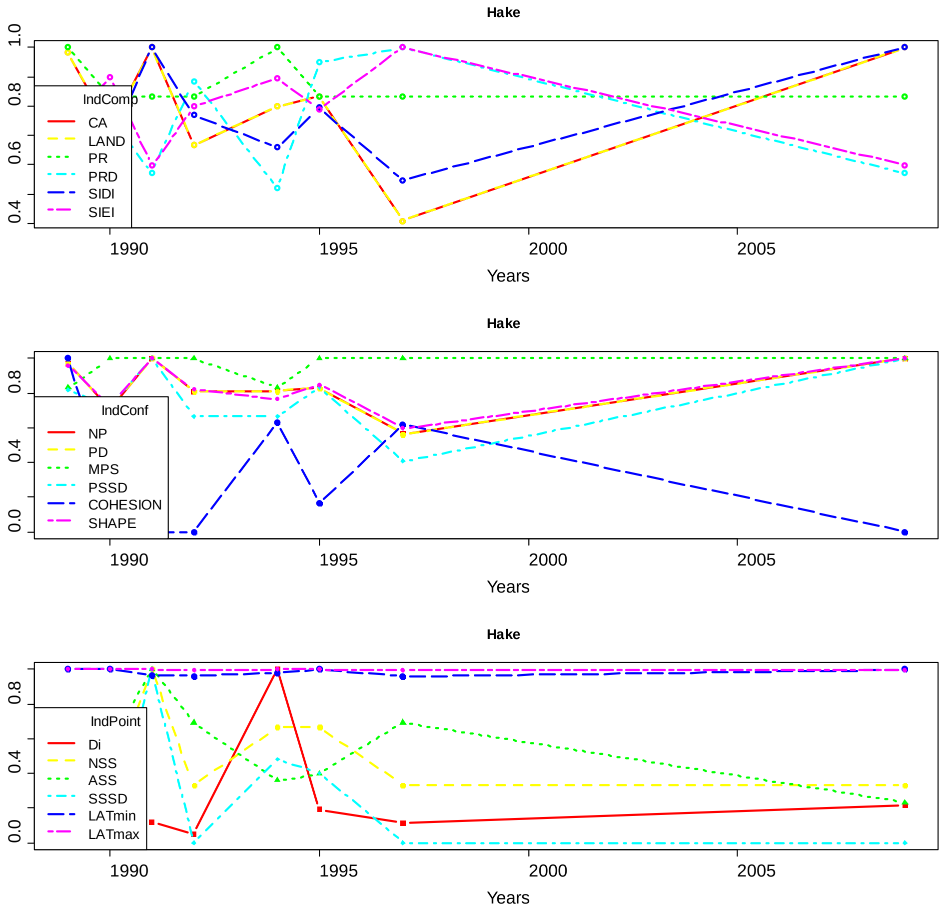

Figure 2.7 (continued). Spatial metrics computed for *Merluccius merluccius* in the Bay of Biscay between 1988–2009 using the EVHOE dataset. No data analysed for 2001–2008.

Using the onboard observations dataset, the spatial metrics could only be calculated for hake as the occurrences of the two other species were too low for metric computation (i.e. two or three values over the complete area under study, see Figure 2.6). Although, this is clearly a limit of the method, any dataset comprising only two values would be anyhow useless. Spatial metrics were computed for the hake even though the time series included only four years. They were either stable across the four years or displayed a slight increase in some of the metrics (e.g. CA, LAND, and Di).

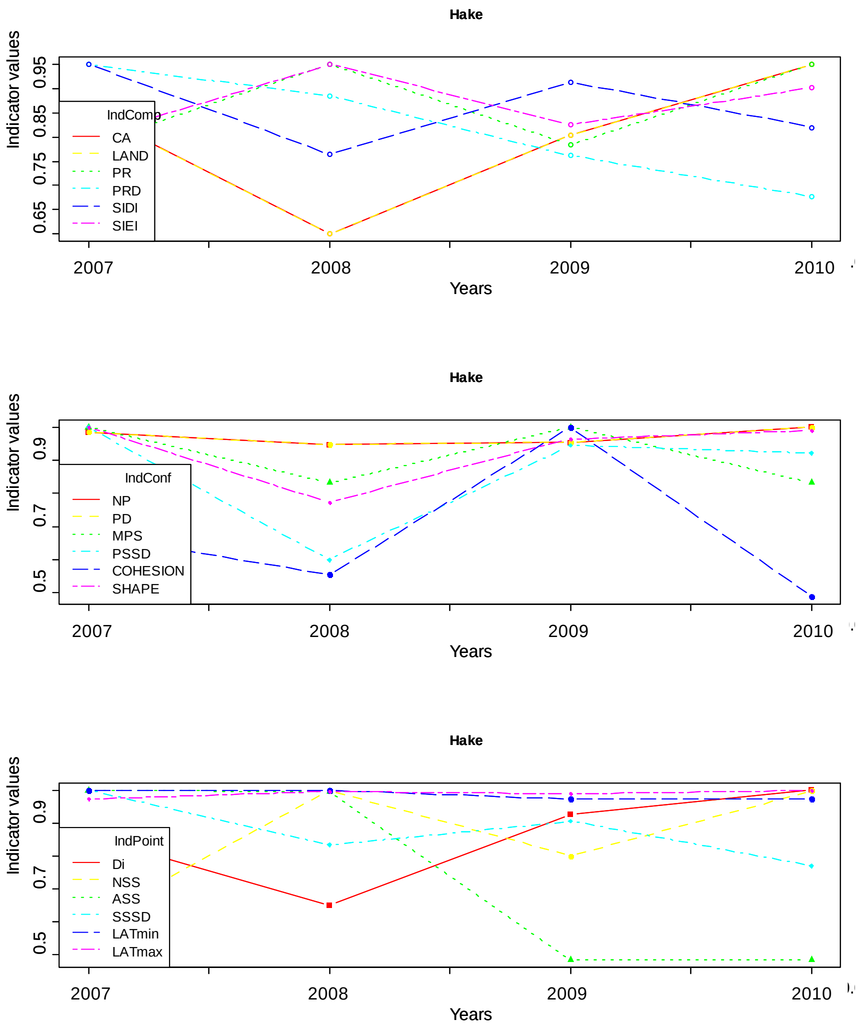

Figure 2.8. Spatial metrics computed for *Merluccius merluccius* in the Bay of Biscay between 2007–2010 using the dataset from the onboard sea observation program (OBSMER).

Barents Sea survey

The Arctic species used in the analyses displayed four contrasted spatial distributions in the Barents Sea (Figure 2.9). *Amblyraja hyperborea* showed low densities homogeneously distributed, *Boreogadus saida* displayed a north-eastern concentration, *Triglops nybelini* was more abundant in the middle-north of the area, and *Aspidophoroides olriki* was restricted to the eastern part of the Barents Sea.

Computation of the spatial metrics for the four Arctic species did not show any clear trends over the studied period (Figure 2.10). There was thus a certain stability for many of the metrics assessing the spatial configuration of the patches and the point-based metrics. This might be due to the short duration of the time series (6 years) or to the fact that the years have all been very warm. Another explanation might be related to the way the time series was created. Maximisation of the common sampling points was done by interpolating the missing values, thereby removing spatial variability from the dataset.

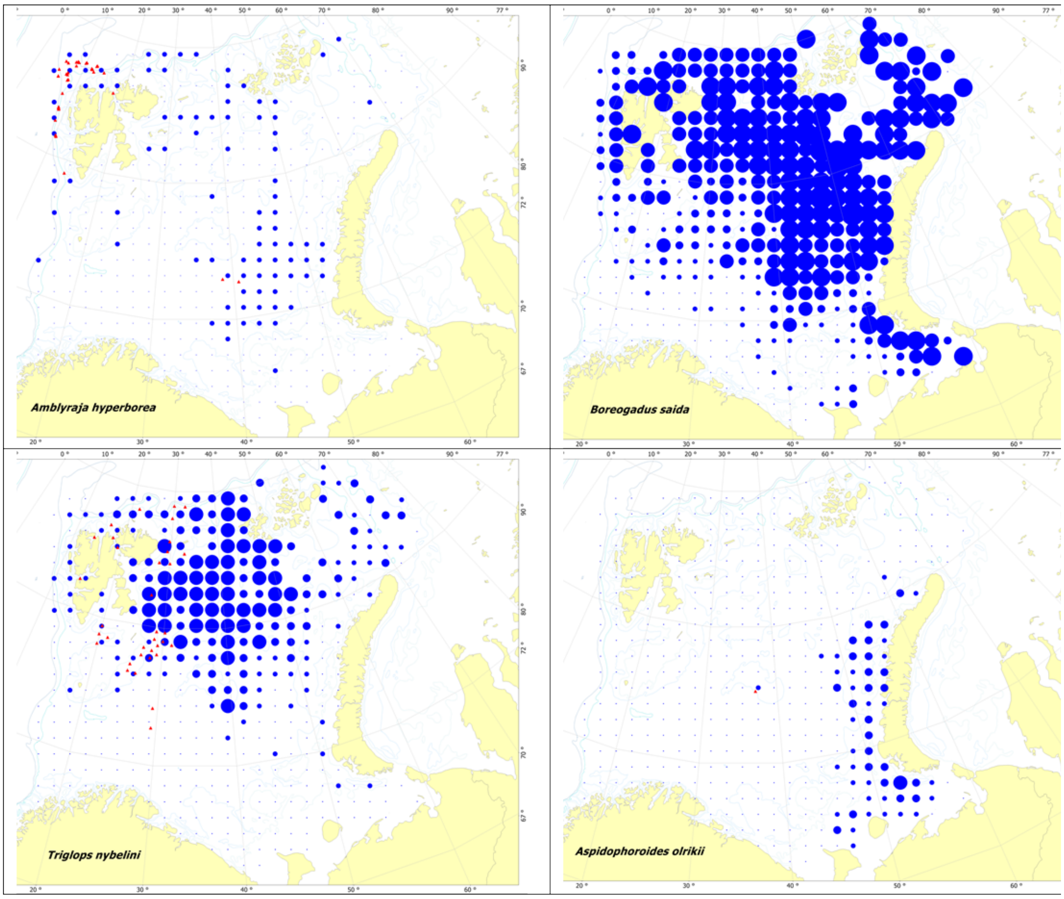

Figure 2.9. Distributions of *Amblyraja hyperborea* (top left), *Boreogadus saida* (top right), *Triglops nybelini* (bottom left) and *Ulcina (Aspidophoroides) olriki* (bottom right) in the Barents Sea 2004–2009. The size of the blue bullets is proportional to the average number of individuals caught per nautical miles towed by 35 by 35 nm grid cells over the six years. The red triangles are catches of specimens identified by experts on fish taxonomy.

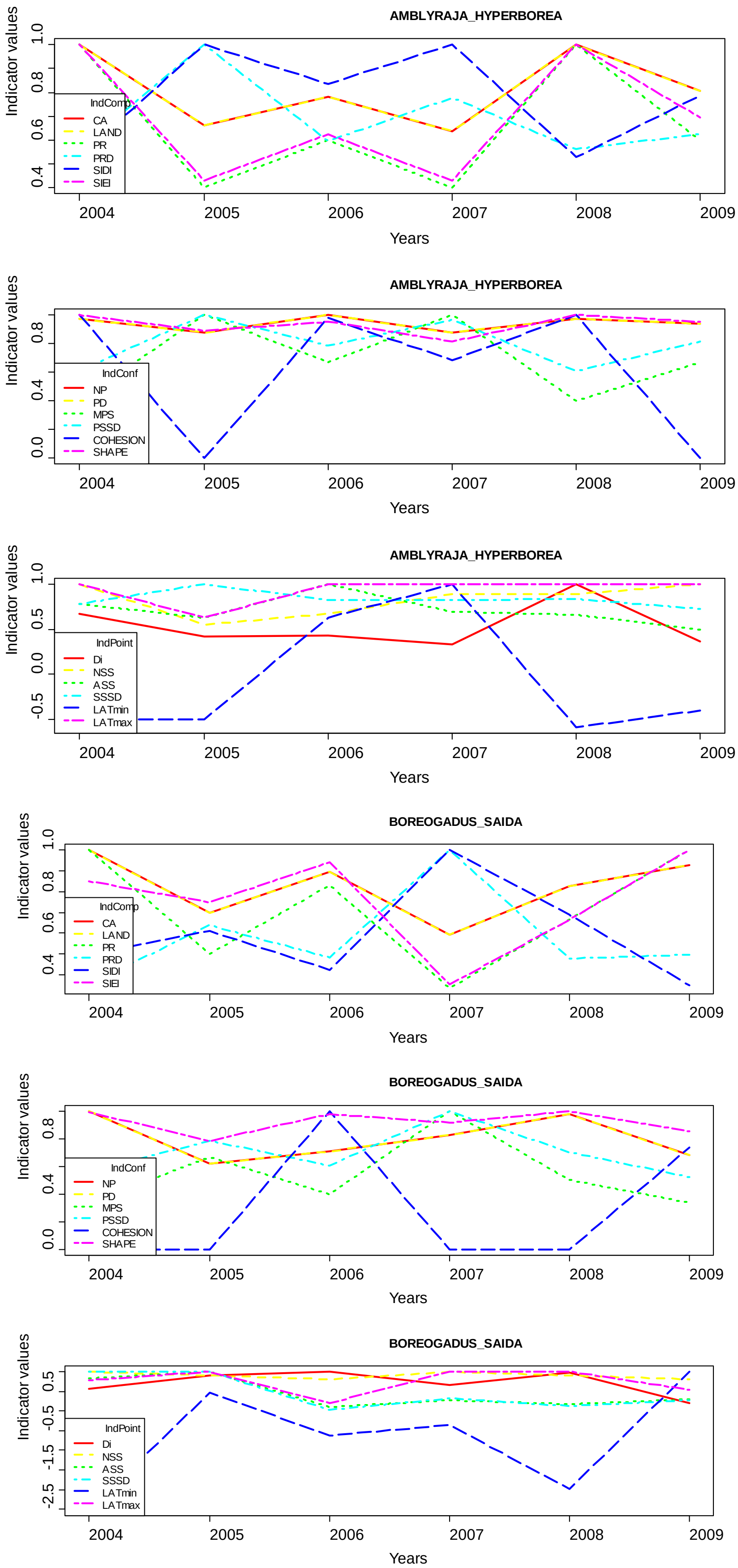

Figure 2.10. Spatial metrics computed for *Amblyraja hyperborea* (top) and *Boreogadus saida* (bottom) in the Barents Sea from 2004 to 2009.

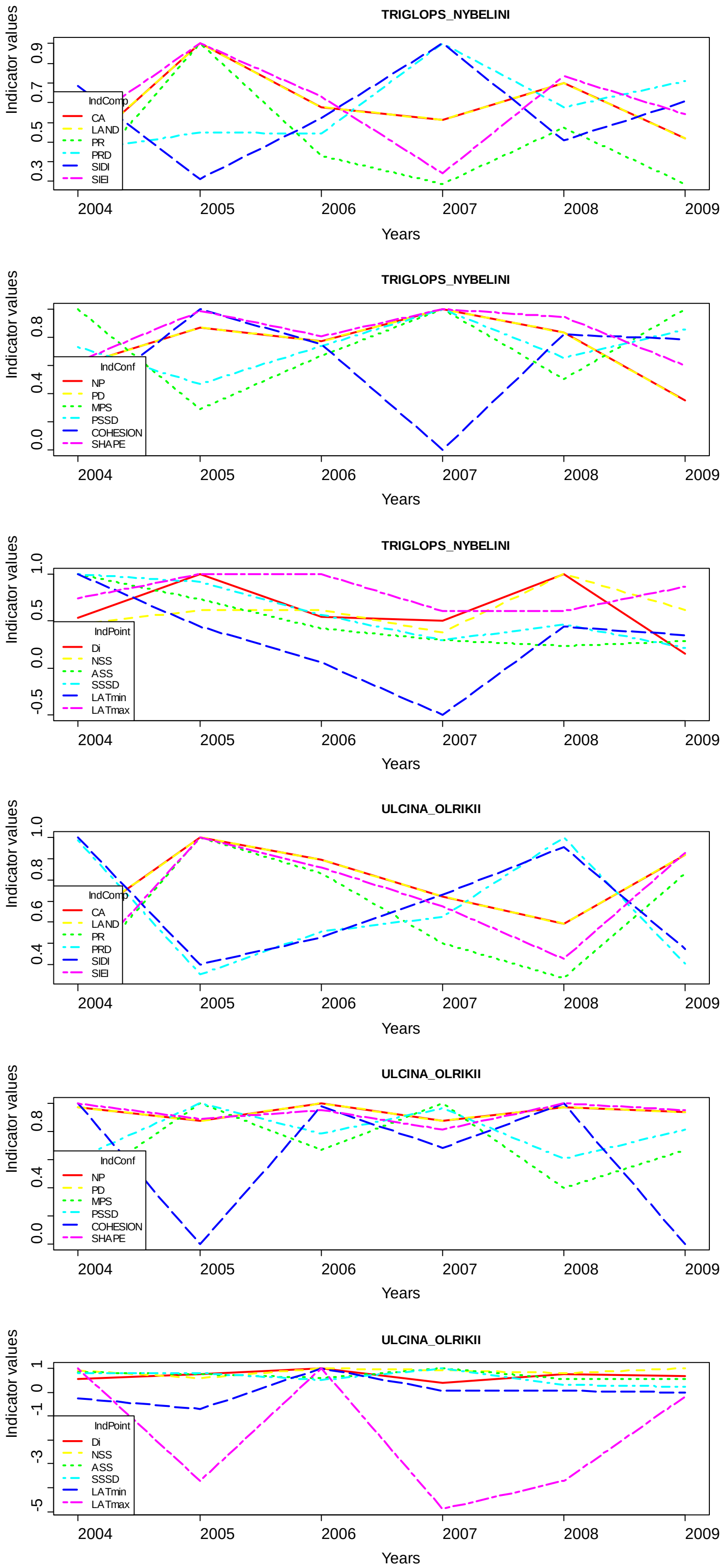

Figure 2.10 (continued). Spatial metrics computed for *Triglops nybelini* (top), and *Ulcina (Aspidophoroides) olriki* (bottom) in the Barents Sea from 2004 to 2009.

Conclusion: This case study was a preliminary examination aiming at defining the methodological limits and constraints of some spatial metrics commonly used in landscape ecology. Several spatial metrics from the three categories (spatial composition, spatial configuration, point-based) displayed trends for some studied species, it is however difficult to interpret them given that the time series are relatively short. Therefore, the results are discussed from a methodological perspective rather than being ecologically interpreted.

Selection of common sampling points: a sampling design issue

The creation of a spatial grid maximising the number of sampling points common to the maximum number of the years had to be done to fix the grain and the sample size of the study over the entire survey period. It generated short time series covering a low proportion of points in comparison to the original number of sampling points each year. In the EVHOE dataset, for instance the selected sampling points created a time series of 8 years, thereby covering 32% of the original duration of the survey's time series. The grid also maximised the number of common sampling points which represented at best 41% of the total sampling points (in 2009). These values are far lower for the onboard observations dataset for which the selected grid generated a times series of 4 years (out of 8 years) and an average proportion of <10% of the total sampling points over the selected years. These low values are due to the observation protocol which by nature is highly variable among years as the "sampling" effort is totally focused on the distribution of specific commercial species and thereby follows their yearly distribution.

A different method was used for the Barents Sea dataset where interpolation was performed in order to keep the maximum number of sampling sites among the years. Interpolation may however damper the spatial variability in the years of low sampling effort, thereby influencing the spatial metrics by homogenizing the spatial patterns. A comparative study will be required to evaluate the effects of this homogenization on the spatial metrics and in particular their time trends.

Systematic sampling with random variation in grid spacing where the same cell is visited each year would have prevented all the data manipulation conducted in this study and would have assured the same sample size among years without any loss of information. Grid sampling designs are very efficient for the description of spatial pattern as they survey the area of interest with the maximum coverage for the minimum effort. Using common grain and extent across years guarantees that the variability observed in the metrics is mainly due to the spatial patterns under study and not to some scaling characteristics. Thus, in the present study, the random or randomly-stratified sampling put severe constraints (small sample size and short time series) on the computation and interpretation of the patch-based metrics.

Sensitivity of patch-based metrics to species distribution maps

Landscape or seascape metrics have been exclusively used for sessile species defining clearly distinct marine habitats. Computation of such indices on mobile organisms was sometimes done in the terrestrial realm but still very scarce in the marine environment (Drew & Eggleston 2008, Meynecke *et al.*, 2008). There are several questions that need to be addressed and some ecological rational to be developed prior to the transfer of these spatial metrics to the marine realm.

One of the most challenging parts of the terrestrial to marine transfer is the development of fish distribution maps defining clear and distinct boundaries. In the present study we used constrained clustering algorithms although several other methods could have been used (e.g. geostatistical analyses, home range analyses and habitat mapping). It would be of prime importance in the future to test other methods to assess the sensitivity of the metrics in regard to the chosen method. However, when categorical maps already exist (e.g. bio-sediment maps, maerl distribution maps, phytoplankton distribution, fish eggs distribution maps) these metrics can be straightforward to compute.

Future developments

This work was a preliminary study primarily focusing on the spatial metrics developed in the landscape ecology discipline. It is actually difficult to state the relevance of these metrics in the frame of the MSFD given the short time series. Other spatial metrics could have been used, notably the geostatistical indicators proposed by Woillez *et al.* 2007. Thus, a more general review including a thorough selection of spatial indicators, from the patch-based metrics to the point-based spatial metrics derived from the Moran's Eigenvector Maps (MEM, Dray *et al.* 2006) and the geostatistical indicators (Woillez *et al.* 2007) is now planned.

2.4.5 Case study: Lessons learnt from the broader application of the Large Fish Indicator (LFI)

Greenstreet *et al.* (2011) described the development of the Large Fish Indicator (LFI) in the North Sea. This indicator was developed to support the setting of an Ecological Quality Objective (EcoQO) for the broader "Fish Community" following the decision that size-based indicators were likely to perform best as a "state" indicator within a "Pressure-State-Response" indicator-based management framework (ICES, 2001a; Greenstreet 2008). This decision was based on the application of seven criteria for a good state indicator (ICES 2001b; Greenstreet 2008). Development of the LFI was done with these criteria in mind; particularly to increase the indicator's sensitive to human pressure on the community and to reduce its sensitivity to alternative drivers of change, such as environmental influences. The main considerations were:

- Which species should be included in the suite of species to which the indicator would be applied?
- What was the best length threshold that should be used to define large fish?
- How should the proportion of fish exceeding this threshold be defined: by numbers or by weight?
- What should the indicator target, the EcoQO, be?

Greenstreet *et al.* (2011) concluded that pelagic species should not be included in the calculation of the LFI. Shoaling characteristics of most pelagic species affects catch probabilities in the survey trawl and the short lifespan and high reproductive capacity of many pelagic species makes variation in their abundance much more responsive to short-term environmental variability compared to most demersal species. Thus including pelagic species would reduce the indicators responsiveness to fishing pressure, and increase its sensitivity to environmental fluctuations and sampling noise.

After comparing a number of different length thresholds, Greenstreet *et al.* (2011) concluded that, for the North Sea IBTS, 40 cm provided the best compromise between having a sufficient proportion of the assemblage exceeding the threshold, and reducing the impact of variable sized recruitment cohorts on the numerator part of the calculation. And finally, Greenstreet *et al.* (2011) determined that calculating the metric based on biomass rather than abundance maximised sensitivity of the metric to fishing pressure (the removal of large fish) and minimised its response to recruitment variation (the addition of small fish). The final definition of the LFI was "the proportion by weight of demersal fish exceeding a length of 40 cm".

Fishing mortality was last deemed to be at sustainable levels in the North Sea in the early 1980s. In 1983, at the start of the Q1 IBTS time series, the LFI was 0.3. Analysis of a second much longer, but now discontinued, groundfish survey time series suggested that over the period 1920 to 1980, the LFI in the North Sea had varied around a value of 0.29. An LFI of 0.3 was therefore adopted as the indicator target, the EcoQO, for the North Sea.

As a final note to the development of the LFI in the North Sea, Fung *et al.* (2012) updated the North Sea LFI time series and in the process of this analysis observed a discrepancy between their time series and LFI trend published by Greenstreet *et al.* (2011). An error in the last two years of the Q1 IBTS data set analysed by Greenstreet *et al.* (2011) was found to be the cause of this discrepancy. Greenstreet *et al.* (2012) rectified this error and repeated all their previous analyses. All the main conclusions drawn in the earlier paper were upheld in this reanalysis, and indeed where in the previous analysis the last two years of data were outliers in their forward-looking model, in the new analysis these two data points fell much closer to model predictions.

The North Sea EcoQO project was a pilot study to develop the basis for an ecosystem approach to management (EAM) that could then be used across all OSPAR regions (Johnson 2008; Heslenfeld and Enserink, 2008). There was therefore a clear need to determine and use the LFI to support an EAM in other marine regions. Shephard *et al.* (2011) therefore developed a LFI for the Celtic Sea. Rather than simply applying the metric determined for the North Sea and adopting the same management target for the indicator, the same EcoQO, they instead followed the same process used in the North Sea to develop their own bespoke Celtic Sea LFI and EcoQO. They again determined that calculating the indicator on the basis of biomass rather than numbers produced a metric with the higher signal to noise ratio, but everything else differed. The suite of species to which the indicator was applied was different. The optimum length threshold was 50 cm and not 40 cm, and the target was set at an LFI value of 0.4 instead of 0.3.

The main conclusion to be drawn from the North Sea and Celtic Sea experiences is that developing indicators and setting targets for ecosystem components across a number of different marine ecosystems is not a case of "one size fits all". It is the concept underlying the construction of particular indicators and the setting of their targets that needs to be consistent across marine regions, not the specific indicators and targets themselves. Thus, indicator definitions and targets may vary from region to region (or from survey to survey), but if the underlying concepts have been applied consistently, then regional integration should be relatively straightforward.

2.4.6 Case study: Utility of the mean maximum length of all fish observed in trawl surveys

There is a reasonable theoretical rationale for considering the maximum size of fish in various assemblages and for species of interest. Human activities (e.g. fishing impacts) could reduce the upper size range of the population, and so the largest maximum observed length of fish may decline in response to fishing. Consideration for such a metric is also in line with the EC decision document (EC, 2010) in terms of the following criteria:

- Population demographic characteristics (e.g. body size or age class structure, sex ratio, fecundity rates, survival/ mortality rates) [1.3.1];
- Composition and relative proportions of ecosystem components (habitats and species) [1.7.1];
- Mean maximum length across all species found in research vessel surveys [3.3.2].

Here we provide a few examples of some of the problems that will need to be overcome before metrics for such criteria can be used in exploratory analyses. Comprehensive analyses cannot be undertaken for some trawl surveys, for example the data held on DATRAS still contains many errors and inconsistencies that need to be addressed. For example:

- Incorrect species: There are still records of species such as *Acentronura, Lamprididae, Leucoraja lentiginosa, Thunnus thynnus, Zenopsis ocellata, Ophidion barbatum, Mullus barbatus, Mugil cephalus, Alosa agone* that will relate to different species (i.e. they are records of misidentified fish or that have been reported using an incorrect species code);
- Inconsistent taxonomy: There are many taxa included in the database under the current, valid scientific name, and also under incorrect synonyms (including junior synonyms and incorrect spellings, Table 2.5);
- Inconsistent reporting procedures for reporting cephalopods and shellfish mean that extensive data filtering is required;
- Incorrect length measurements due to incorrect allocation of units (mm/cm), especially in terms of shellfish, clupeiforms and sandeels;
- Incorrect length measurements due to incorrect species identifications. Even a **brief examination** of DATRAS data identified many species for which the $L_{max. obs.}$ was much greater than the L_{max} that would be expected from the literature (Table 2.6).

Table 2.5. Examples of taxonomic inconsistencies and spelling errors in DATRAS data.

Scientific name	Junior synonym	Spelling mistake
Leucoraja fullonica	*Raja fullonica*	
Leucoraja naevus	*Raja naevus*	
Amblyraja radiata	*Raja radiata*	
Dipturus batis	*Raja batis*	
Psetta maxima	*Scophthalmus maximus*	
Solea solea	*Solea vulgaris*	
Maja brachydactyla	*Maja squinado* *Maia squinado*	*Maia squinado*
Entelurus aequoreus		*Entelurus aequerius*

DATRAS data, especially for non-commercial species, will require thorough data checking and quality assurance before it can be used to create a reliable metric to examine the '*Mean maximum length across all species found in research vessel surveys*'. Available data may be suitable to examine the largest observed individuals over time for selected species (i.e. for a range of fish for which identification problems are not problematic). Example plots of the trends in $L_{max. obs.}$ for selected species is illustrated in Figure 2.11. These plots should be viewed as illustrative only, as the data quality cannot yet be assured.

For many species in Figure 2.11 $L_{max.\ obs.}$ has decreased over the last decade. However, $L_{max.\ obs.}$ was highly variable between years and the negative trends are not always consistent within a species across quarters (Q1 and Q3). The decline of $L_{max.\ obs.}$ was most evident for haddock, tusk and wolf-fish. For anglerfish and ling an increase in $L_{max.\ obs.}$ occurred after 2005. Given that annual values can be highly variable, the $L_{max.\ obs.}$ as an indicator of population condition should be viewed over appropriate (long-term) time-scales. Annual fluctuations in $L_{max.obs.}$ may be attributable to the random effects of caching a large individual as much as to true changes in the length structure of the population. Hence only longer-term patterns may be informative of the loss of larger individuals, although they may usefully support other population/assemblage indicators.

The misclassification of closely related species may cause inconsistencies in the time-series, as thought to be seen for *Raja clavata*. In 1994 (Q3), for example, the $L_{max.\ obs.}$ fell, however, during this year specimens of *A. radiata* > L_{max} were reported, which could have been missidentifed *R. clavata*. This emphasizes the weakness of metrics which are based on measurements of single individuals, especially when data quality is uncertain.

Table 2.6. Examples of where L_{max} (from DATRAS) is greater than expected from the scientific literature.

Quarter	Scientific name	Lmax. obs. (mm, DATRAS)	$L_{max.}$(mm, Wheeler, 1978)
Q1	*Agonus cataphractus*	260	207
	Ammodytes marinus	370	240
	Ammodytes tobianus	350	200
	Callionymus lyra	380	300
	Callionymus maculatus	300	140
	Engraulis encrasicolus	1350	200
	Leucoraja naevus	1060	700
	Pholis gunnellus	370	250
	Raja montagui		750
	Spinachia spinachia	400	200
Q3	*Taurulus bubalis*	300	175
	Squalus acanthias	1480	1220
	Leucoraja naevus	980	700
	Amblyraja radiata	1380	760
	Pomatoschistus	190	95
	Gymnammodytes semisquamatus	1500	235
	Echiichthys vipera	530	140
	Callionymus maculatus	560	140

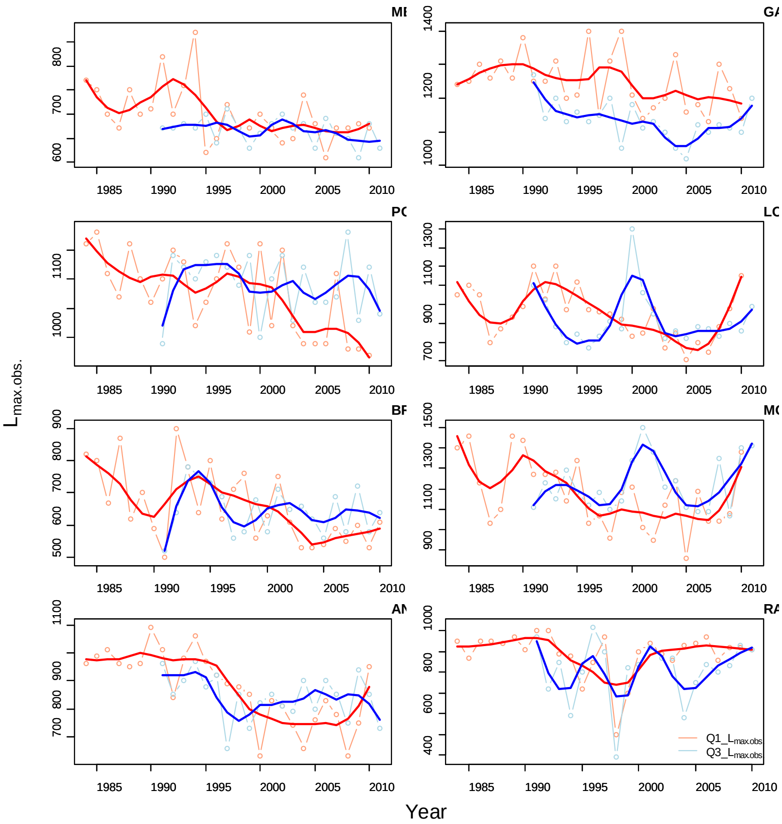

Figure 2.11. Time-series of maximum observed length ($L_{max.obs.}$) for eight North Sea fish species. Data from IBTS quarter 1 (red) and quarter 3 (blue). Solid lines are loess-splines of time-series. Shown for illustrative purposes only, as there may be erroneous data for L_{max} in DATRAS.

EC (2010) suggested the use of the "*mean maximum length across all species found in research vessel surveys*" (3.3.2) as an indicator for the status of exploited fish populations in Descriptor 3. However, because Descriptor 3 generally deals with the health of exploited fish populations, WGBIODIV considered that mean L_{max} would be of relevance to Descriptor 1, criteria 1.7, which addresses the state of assemblages of taxonomic groups.

In the time-series of the North Sea IBTS from 1984 to 2010, all species (N=49) for which a $L_{max.obs.}$ was available within the full time-series of 27 years were used to calculate a mean community $L_{max.}$. The resulting $L_{max.}$ was compared to the Large Fish Indicator (LFI) based on Greenstreet *et al.* (2011; Figure 2.12).

Both community indicators displayed a declining trend, but were not strongly correlated to each other. It may be questioned whether the LFI and the L_{max}. express the same properties of the fish assemblage. The LFI expresses biomass and hence is more affected by the catch volume of large fish, whereas the mean L_{max} is independent of catch mass. Weighting the mean $L_{max.}$ by survey CPUE may lead to different outcome. Furthermore, a full quality assessment with regards to sensitivity, responsiveness and specificity of the mean $L_{max.}$ remains to be performed (see Rice and Rochet, 2005).

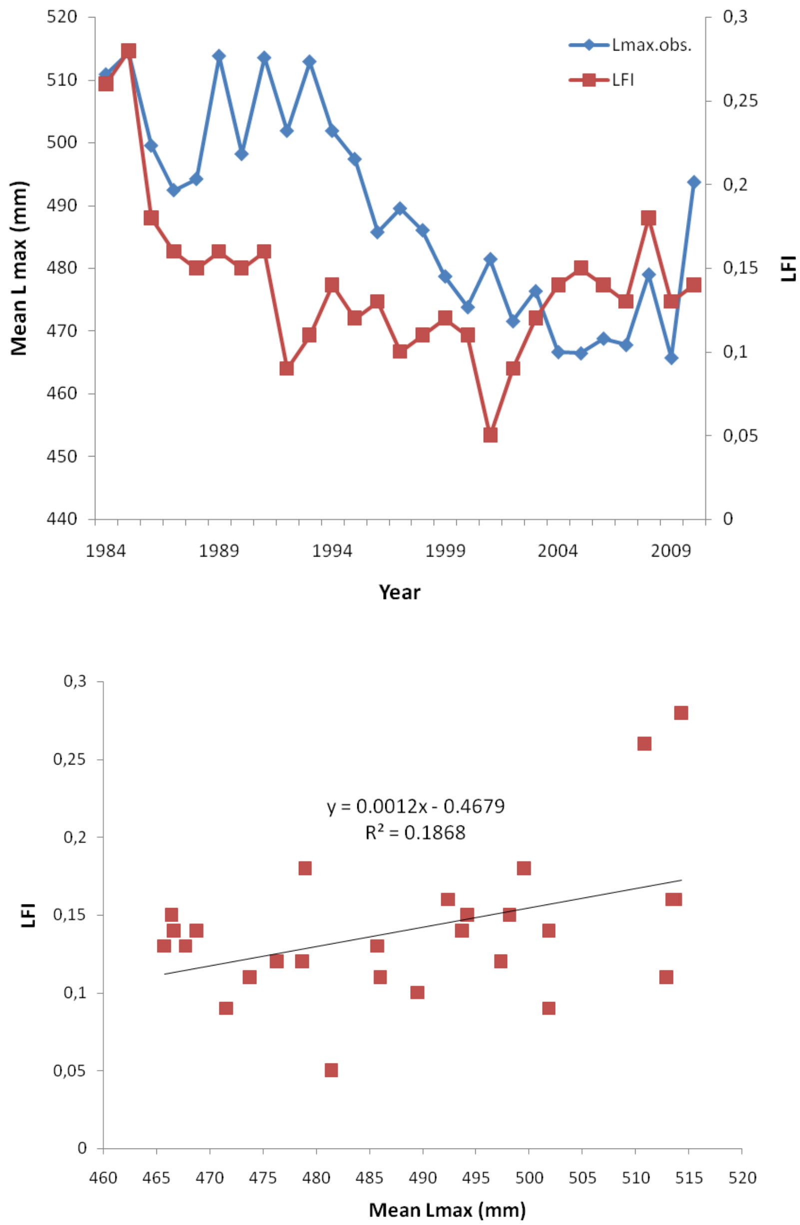

Figure 2.12. The mean L_{max}. and the LFI from 1984–2010 (top) and correlation between the LFI and mean L_{max}. (bottom).

2.4.7 Appropriate geographical scales for case study species

WGBIODIV were asked to comment on the appropriate geographic scales for ensuring trans-boundary species/habitats are assessed at biologically-meaningful scales. Here we provide an example case study highlighting the utility of current fishery-independent surveys to sample the coastal skate species *Raja undulata*.

Undulate ray *R. undulata* is a large-bodied and patchily distributed elasmobranch that is currently listed as a 'prohibited species' on the EC's TAC and quota regulations. Hence, such a species may be considered by relevant Member States as a candidate species for biodiversity monitoring at some stage.

Data from existing IBTS and beam trawl surveys and other sources highlight that this species is only taken in selected areas within the English Channel and parts of the Celtic Seas and Biscay/Iberia eco-regions (Ellis *et al.*, 2012). Furthermore, although ichthyological guides suggest that the bathymetric range extends down to 200 m, data analyses highlight that this species is usually encountered in waters of <50 m deep and often in coastal waters of <20 m deep (Figure 2.13). Given that such inshore grounds are not sampled comprehensively by many existing fishery surveys, quantitative data for such a species will be lacking for much of the distributional area in the absence of dedicated coastal surveys or fishery-dependent information.

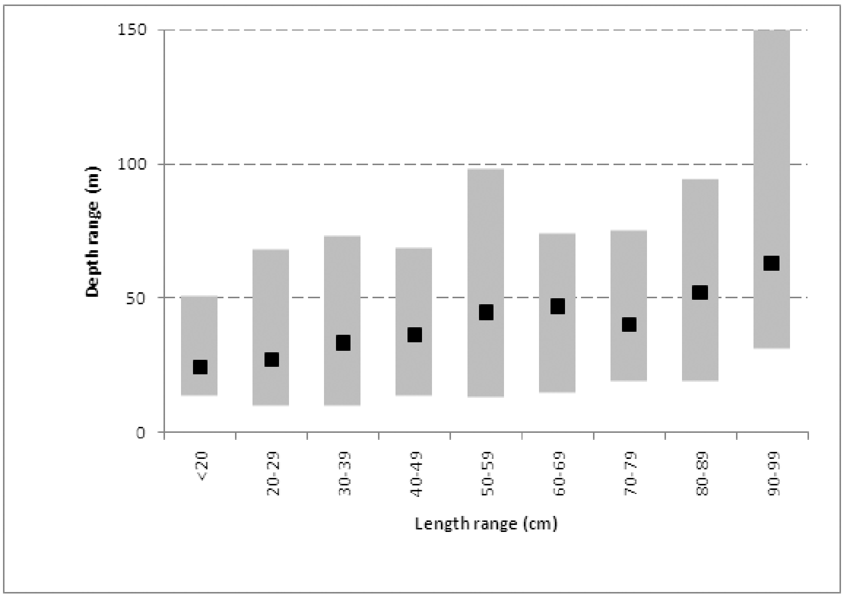

Figure 2.13. Bathymetric distribution of records of *Raja undulata* (as recorded in recent IBTS and beam trawl surveys in the north-east Atlantic) by length class, giving the mean depth (black squares) and minimum and maximum depths at which fish in the size category were observed (grey bars). (Source: Ellis *et al.* 2012).

2.5 Regional integration: HELCOM's integration of indicators

This section is largely based on Korpinen & Zweifel (2012).

HELCOM's assessments have taken place in 1986, 1990, 1996, 2001 and 2003 (Korpinen & Zweifel 2012). In recent years this work has been influenced by the Baltic Sea Action Plan (BSAP, adopted in 2007) and more recently by the MSFD.

HELCOM CORESET is a project for the development of core indicators for hazardous substances and biodiversity (started in June 2010) and this work is tightly coupled with the HELCOM BSAP and the MSFD. A core indicator makes it possible to assess the current status and also allows for the follow-up of the progress for achieving GES as described in the MSFD. These indicators also make it possible to measure the progress towards HELCOM's ecological objectives and goals.

The interim report (Korpinen & Zweifel 2012) describes the underlying principles of the work (Table 2.7), the process of selecting core indicators, the justifications for the choices made and it also identifies gaps.

Table 2.7. Common principles for HELCOM core indicators. Source: Korpinen & Zweifel (2012).

No.	Principle	Description
1	Compiled and updated by Contracting Parties	
2	Science-based	Each indicator describes a scientifically sound phenomenon.
3	Link to anthropogenic pressures	Status indicators should be linked to anthropogenic pressures and indirectly reflect them, where appropriate, and additional pressure indicators are used and they directly reflect anthropogenic pressures and are tightly linked to human activities.
4	Policy response	The indicator measures part of or fully an ecological objective and/or a descriptor of good environmental status.
5	Suitability with assessment tools	The indicator can be used with the assessment tools but the assessment tools will be open for modifications as necessary.
6	Suitability with BSAP/MSFD, making best use of the synergies with other Directives and according to the HELCOM Monitoring and Assessment Strategy	The indicator reflects a component contained in the HELCOM system of the vision, goals and ecological objectives and/or MSFD descriptor.
7	Qualitative or quantitative with a textual background report	Indicators (qualitative or quantitative), are numeric, based on measurements or observations and validated models; they must also have a quantitative target level reflecting the lowest boundary of GES. They also contain a textual background report with interpretation of the indicator results. The report should be published on the HELCOM website and ultimately should take the form of the three-layered indicator report (cf. preliminary core eutrophication indicator reports) with the main page containing a status map and the main message aimed at decision makers; the second page containing trend information, and the third page containing technical background information and information on the confidence of the assessment.
8	Baltic Sea wide	The HELCOM indicators should cover the whole sea area.
9	Commonly agreed	The finalised indicators and their interpretation are commonly agreed among the HELCOM Contracting Parties and HELCOM MONAS is the HELCOM body that should approve the publication of the core indicator reports on the HELCOM website.
10	Frequently monitored and updated	Data underlying the indicators are collected within the HELCOM coordinated monitoring (HELCOM COMBINE, MORS-PRO, PLC) and the indicator reports will be updated preferably annually or at intervals suitable for the measured factor.
11	Harmonised methodology	Data in an indicator will be collected using harmonised monitoring, quality assured analytical methods, as well as harmonised assessment tools, according to the relevant HELCOM guidelines or EU standards, such as methodological standards or guidelines for GES under the MSFD to be delivered by the EC, other relevant international standards.
12	Confidence evaluation	The indicator and the data must be assessed using common criteria and this confidence evaluation is to be included in the indicator report.

The interim report also provided a framework for indicator selection that reflected the impact of the main anthropogenic pressures on selected key species, functional groups and predominant habitats, although it does not yet cover all ecological objectives of the BSAP or descriptors of the MSFD (because the work is still in progress). The chosen core indicators were narrowed down from a list of several hundred potential indicators. At present 15 core indicators have been chosen for biodiversity assessments with an additional 23 supplementary indicators (Tables 2.8 and 2.9).

Of the fourteen core indicators, eight describe species and five describe habitats. No core indicators for the genetic structure of populations and the distributional patterns of habitats have been developed, mainly due to information and monitoring gaps. Species distribution is only assessed with a single indicator (seabirds) although similar information should be available for other groups. Similarly, the ecosystem structure is only described by the trophic index of the coastal fish assemblage. Marine mammals, seabirds and fish are relatively well represented as core indicators. The most obvious gaps are the lack of indicator proposals related to underwater habitats and several of the key functional groups or species in the Baltic Sea. Further work to fill these gaps is now in progress. Several of the listed candidate indicators are expected to be developed into core indicators before the end of the project (Korpinen & Zweifel 2012). Targets will be developed for all indicators according to Table 2.10.

Table 2.8. Proposed core indicators for Descriptor 1 at the species level. The core indicators are shown in their relation to MSFD GES criteria and indicators (EC Decision 477/2010/EU). Source: Korpinen & Zweifel (2012).

Criteria	Type of indicator	Proposed indicator
1.1 Species distribution	1.1.1 Distributional range	Birds: Distribution of wintering seabird populations the latter
	1.1.2 Distribution pattern within	No indicator as yet
	1.1.3 Area covered by sessile/benthic species	No indicator as yet
1.2 Population size	1.2.1 Abundance and/or biomass	Mammals: Population growth rate of marine mammals
		Fish: Fish population abundance
		Birds: Abundance of wintering populations of seabirds
1.3 Population condition	1.3.1 Population demographic characteristics: (body size or age class structure, sex ratio, fecundity rates, survival/ mortality rates)	Mammals: Blubber thickness of marine mammals; pregnancy rate of marine mammals
		Birds: White-tailed eagle productivity
		Fish: Mean metric length of key fish species
	1.3.2 Population genetic structure	No indicator as yet

Table 2.9. Proposed core indicators for the Descriptor 1, habitat level (including associated communities) and ecosystem level. The core indicators are shown in their relation to MSFD GES criteria and indicators (EC Decision 477/2010/EU). Source: Korpinen & Zweifel (2012).

Criteria	Type of indicator	Proposed indicator
1.4 Habitat distribution	1.4.1 Distributional range	No indicator
	1.4.2 Distributional pattern	No indicator
1.5 Habitat extent	1.5.1 Habitat area 1.5.2 Habitat volume	Seabed communities: Lower depth distribution limit of macrophyte species
1.6 Habitat condition	1.6.1 Condition of the typical species and communities	Seabed communities: Multimetric macrozoobenthic indices (e.g. BQI, MarBIT, DKI, BBI)
		Fish: Proportion of large fish in the community; Fish community diversity
	1.6.2 Relative abundance and/or biomass	Fish: Abundance of fish key trophic groups
	1.6.3 Physical, hydrological and chemical conditions	Water transparency; Inorganic N; Inorganic P; Chl a
1.7 Ecosystem structure	1.7.1 Composition and relative proportions of ecosystem components	Fish community trophic index

Table 2.10. Common principles for quantitative or qualitative targets of core indicators. Source: Korpinen & Zweifel (2012).

1	Targets need to be developed for each indicator separately.
2	**Purpose of the status targets**: The target reflects the boundary between GES and sub-GES. The boundary can be based on a specific score (cf. ecological quality ratio, EQS, *sensu* WFD and also used in HEAT and BEAT) that can be derived through the use of an 'Acceptable deviation' from a 'Reference condition'.
3	**Purpose of the pressure targets**: The targets reflecting anthropogenic pressures should guide the progress towards achieving good environmental status.
4	**Science-based**: A target level should be based on best available scientific knowledge. In the absence of data and/ or modelling results, expert judgment based on common criteria should be involved to support the target setting.
5	**Spatial variability**: Target levels can vary among sub-basins or among sites depending on natural conditions.
6	**Confidence of the targets**: These must be evaluated by common criteria and included in the general confidence evaluation of the indicator report.

The HELCOM CORESET project was also tasked to develop qualitative descriptions of the GES boundaries of the core indicators and for some of these the descriptions are developed for the criterion level aimed to facilitate the development of quantitative GES boundaries (Korpinen & Zweifel 2012). Table 2.11 shows some examples of these qualitative descriptions as examples of the approach used by HELCOM CORESET.

One of the basic features of the HELCOM core indicators is that they reflect anthropogenic pressures on a species, community or habitat, or that they follow the intensity of the pressure directly. The HELCOM CORESET, so far, has concentrated on state indicators, but the aim is to shift the work gradually also to pressure indicators.

Table 2.11. Examples for developing qualitative descriptions of GES boundaries for selected state indicators (Source: Korpinen & Zweifel 2012).

GES Criterion	Suggestion for GES description	Approach for setting the GES boundary	Description of the GES boundary
1.5 Habitat extent	Habitat extent (areal extent and/or volume) in line with prevailing physiographic, geographic and climatic conditions; loss of extent is minimized but accommodates defined levels of sustainable use	Lower depth distribution limit of macrophytes: Several approaches as used in the WFD: generally within natural fluctuations of what has been defined as type specific reference conditions	GES is met when the lower depth distribution of macrophytes shows only slight signs of disturbance.
1.6. Habitat conditions	The habitat (defined by abiotic and biotic parameters) is in a condition to be able to support its ecological functions and the diversity of its associated community.	Macrozoobenthos indices: Several approaches as used in the WFD	GES is met when the level of diversity and abundance of invertebrate taxa only slightly outside the range associated with type-specific conditions.
		Community size index coastal fish: Based on site-specific reference data. If reference data are missing, trends in available data and expert judgements are used to assess the status	GES is met when the size structure of the fish community is at an appropriate level to support community function (inc. food provision and resilience). It is given as ind. >X cm and is primarily site-specific.
		Community diversity index coastal fish: as above	GES is met when the diversity of the associated fish community is at an appropriate level to support community function and resilience; is based on reference data series that represents GES; is given as a unitless index value >X>; and is primarily site-specific.
		Community abundance index coastal fish: As above	GES is met when the abundance of cyprinids and piscovores is at an appropriate level to support the community functions and resilience, based on the reference data series that has been defined as representing GES; is given as abundance >X>; and is primarily site –specific.
		Biomass of copepods (absolute or relative): GES is based on a reference data set that represents a time period when zooplanktivorous fish growth/condition and fish stocks were relatively high	GES is met when the copepod biomass is sufficient to support favourable feeding conditions for zooplanktivorous fish. A GES boundary is defined for each sub-basin of the Baltic Sea provided that monitoring data for the area are available.
		Biomass of microphagous mesozooplankton (absolute or relative): GES is based on a reference data set that represents a time period when cholorphyll and water transparency complied with GES	GES is met when the biomass of microphagous mseozooplankton does not exceed levels typical for the Baltic Sea unaffected by eutrophication. A GES boundary is defined for each sub-basin and WFD coastal water types.

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3 Biodiversity indicators linked to ecosystem function

3.1 Introduction

WGBIODIV were given the ToR (d) to "Review and report on existing indicators of biodiversity that are linked to predictable changes in ecosystem function and/or to develop, assess and report on the feasibility and performance of such indicators".

Indicators of biodiversity, as suggested under the MSFD, are reviewed in the previous section. WGBIODIV had a ToR in 2010 to "Review existing approaches to the development of biodiversity indicators", and so provided on overview of metrics for species diversity (including some species-specific metrics, community metrics, taxonomic diversity metrics, functional diversity metrics, size-based indicators, food-web indicators, surrogate methods, indices of biological integrity) and also considered genetic diversity and habitats and biotopes (see Section 4 of ICES, 2010). Further studies of biodiversity indicators were also undertaken last year (see Section 3 of ICES, 2011). In terms of ecosystem function, these have been discussed in earlier reports of WGBIODIV (see Sections 3 and 4 of ICES, 2010; Section 6 of ICES, 2011). A brief overview of linking biodiversity metrics to ecosystem in given below.

3.2 Biodiversity indicators

A wide range of biodiversity metrics are available, and several of these may allow temporal and spatial patterns in the ecological structure and function of assemblages and ecosystems to be assessed (Gallardo *et al.*, 2011), or for developing species-specific indicators for taxa that are considered functionally important (e.g. foundation species, keystone species etc.).

However, such metrics can vary considerably in costs (for surveys, processing and analyses), variability, sensitivity to predictable changes in specific activities and how closely they are linked to changes in ecosystem function. Few indicators have been quantitatively tested to determine whether or not they meet the criteria for their selection (Heink & Kowarik, 2010; Tables 3.1 and 3.2), and comparable criteria would also apply to indicators of biodiversity that are linked to ecosystem function.

Table 3.1. ICES criteria for a good indicator (adapted from ICES, 2004).

Criterion	Property
A	Relatively easy to understand by non-scientists and those who will decide on their use
B	Sensitive to a manageable human activity
C	Relatively tightly linked in time to that activity
D	Easily and accurately measured, with a low error rate
E	Responsive primarily to a human activity, with low responsiveness to other causes of change
F	Measurable over a large proportion of the area to which the metric is to apply
G	Based on an existing body or time-series of data to allow a realistic setting of objectives

Table 3.2. Criteria for a good indicator (adapted from Normander *et al.*, 2012).

	Quality	Description
1	Representative and with good coverage	Includes a sufficiently large and representative group of species and covering an appropriate and meaningful spatial extent
2	Temporal coverage and up-to-date	Shows temporal trends and can be updated routinely (e.g. annually) with existing surveys
3	Simplifying information	Summarises data into a simple form that is intelligible to wider society
4	Clear presentation	Possible to display information in clear graphical representation
5	Indicative	Indicates meaningful changes occurring over an appropriate scale
6	Sensitive to anthropogenic change	Measured qualities sensitive to changes due to the management of anthropogenic activities and less influenced by natural variability and fluctuations in the environment
7	Quantitative and statistically sound	Based on quantitative observations and from statistically sound survey methods

8	Relatively independent of sample size	Usable data may be obtained with relatively small sample sizes
9	Realistic	Based on existing monitoring programmes or, if requiring further monitoring, can be brought in cost-effectively and are economically possible
10	Acceptable to stake holders	Responds to the needs of stakeholders and are generally accepted by them
11	Policy relevant	Linked to clear policy requirements and so can assessing progress towards national or international targets
12	Can be explained	The impact and significance of any changes measured by the metric should be known
13	Predictable	Theoretical predictions should be available, so that it can be linked into forecast models
14	Comparable	Enables comparison (e.g. between nations)
15	Can be aggregated/disaggregated	Data may be aggregated and disaggregated into different levels (e.g. taxonomically or regionally)

3.3 Ecosystem function

Ecosystem functions are "intrinsic ecosystem characteristics related to the set of conditions and processes whereby an ecosystem maintains its integrity" (Harrington *et al.*, 2010). Ecosystems provides a wide variety of functions (see Section 6 of ICES, 2011), including for supporting services, such as habitat formation and primary production, and regulating services, such as nutrient cycling and energy flow, oxygenation and gas exchange, decomposition, and the biological regulation of populations etc.

Depending on the marine habitat type, geographical area and exact nature of the function, there can be a variable number of species contributing to the function, as a broader number of species contributing to a function should provide some degree of resilience. Hence, the type of indicator that would be selected to inform on a defined ecosystem function would need to be investigated on a case by case basis.

3.4 Examples of linking biodiversity indicators with changes in ecosystem function

Currently, the analyses of links between biodiversity and ecosystem functioning are moving beyond assessments of species richness towards trait-based analyses (e.g. Hooper *et al.*, 2005). Body size is a potentially an important trait (both within and among species) due to its influence on population abundance, geographic distribution, species interactions, life history adaptations, and physiological profiles (Blackburn & Gaston, 1994; Hildrew *et al.*, 2007), and its role in structuring trophic interactions (Jennings *et al.*, 2001; Hildrew *et al.*, 2007; Shackell *et al.*, 2010). At the community level, body sizes can largely determine the types and strengths of flows of energy and materials in ecosystems thereby affecting ecological networks (Daufresne *et al.*, 2009; Woodward *et al.*, 2005).

A recent study in the North Atlantic has shown that a climate-induced increase in marine copepod diversity, with a concurrent shift towards a smaller mean community body size, may have negative effects on the drawdown of biological carbon and on fisheries by influencing the networks through which carbon flows (Beaugrand *et al.*, 2010). Similarly, for fish species, global analyses have indicated that body size may act as an important factor in mediating the relationship between marine fish species richness and ecosystem functioning, suggesting that the management initiative to ensure ecosystem stability in the face of exploitation should focus on this important functional trait (Fisher *et al.*, 2010).

Some ecosystem functions can support higher levels of biodiversity. For example, some studies have shown how there can be an increased diversity and/or abundance of, for example fish, in areas with habitat-forming species, such as coral, kelp, seagrass and oyster reef (Bracken *et al.*, 2007). Further examples of such 'biodiversity hotspots' are discussed in Section 4, and more examples of the role of habitat-forming species in supporting a high diversity and serving as 'essential fish habitat' were given in the reports of the Working Group on Fish Ecology (see Section 4 of ICES, 2003; Section 5 of ICES, 2004; Section 3 of ICES, 2005; Section 5 of ICES, 2006).

Nevertheless, there have been few generic studies evaluating biodiversity indicators that are linked to ecosystem function in marine ecosystems (Danovaro *et al.*, 2008), although there have been a range of such studies in terrestrial and freshwater ecosystems. Further studies to determine the utility of biodiversity indicators for informing on the structure and function of marine ecosystems are required.

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4 Integrating biological data to better describe 'biodiversity hotspots'

4.1 Introduction

This section addresses ToR b "Review and report on methods to integrate biological data from disparate sources and for different components of the ecosystem to better describe biodiversity hotspots".

WGBIODIV reviewed some of the potential methods for combining biodiversity data in a previous report (ICES, 2011), and so this is only addressed briefly in the current section. The concept of biodiversity hotspots, which was mentioned in an earlier report (see Section 3 of ICES, 2010), is more thoroughly discussed below, with some case studies also included to highlight potential uses and issues of survey data to define areas as biodiversity hotspots.

4.2 Biodiversity hotspots

An improved knowledge of biodiversity hotspots in the ICES area is required, as this may help inform on monitoring programmes and analyses of field data, and may also inform on spatial management. Biodiversity hotspots have often been promoted as a cost-effective method of protecting biodiversity, but it has also been argued that species-poor habitats may need "conservation effort in order to avoid the loss of essential ecosystem services" (Price, 2002).

Although the term 'biodiversity hotspot' is used widely, there is surprisingly little consensus on the actual definition (Willis *et al*., 2007). Hiscock & Breckels (2007) defined marine biodiversity hotspots as "areas of high species and habitat richness that include representative, rare and threatened features". Allen (2008) stated that the term hotspot was used "to denote a relatively restricted geographic area containing an extraordinary high concentration of biodiversity and endemism". Other definitions (based on terrestrial ecology) also include that biodiversity hotspots should have high levels of endemism. Whereas some organisations have suggested that biodiversity hotspots should also be sites under threat, other organisations suggest that they should be areas that are largely intact.

WGBIODIV considered that there may be several bases for what may constitute biodiversity hotspots. These may represent sites or regions where there is an increased biomass, density, and/or species richness of one or multiple taxonomic groups, which may be often associated with areas of high productivity (which may be surface or sub-surface). They may also be areas where there is a high level of endemism or a concentration of rarer species and habitats.

Biodiversity hotspots may also include those areas where numerous species aggregate for ecologically important functions (e.g. for feeding, breeding, sheltering, overwintering and migrations). Hence there can be some similarity between biodiversity hotspots with the concepts of ecologically or biologically significant areas (EBSAs) (e.g. see CBD, 2008; DFO, 2011). Some fishery surveys in the ICES area have caught large aggregations of a particular species at on a certain sites, but it is also important to know if the importance of such sites is persistent over time. Sites where there is an abundance of food (e.g. forage fish) that may be exploited by various top predators may also serve as biodiversity hotspots.

On a finer scale, biodiversity hotspots in the benthic and demersal environment may also be influenced by habitat complexity, whether based on finer scale topographic features, (e.g. Miller & Etter, 2011) or presence of structural fauna (e.g. kelp forests, cnidarians, bryozoans, ascidians and other sessile taxa that create complex or reef-like structures) that may support a high diversity of associated fauna.

It should also be noted that areas of high species diversity are not always consistent across taxa. For example, there can be a larger number of nematode species on the crests of sand banks than at off-bank sites, whereas the number of infaunal and epifaunal species is higher on off-bank sites (Ellis *et al*., 2011).

The intermediate disturbance hypothesis suggests that some level of disturbance will maximise diversity, as it maximises the chances of more species to coexist.

As with many aspects of marine ecology, issues of scale are fundamental. Biodiversity hotspots may be identified on a global, regional, or more site-specific scale. They may be variable in space and time. For example, topographical features such as seamounts may act as hotspots, but so can oceanographic features (e.g. upwellings, frontal systems, eddies), which may be more variable in location and timing.

It has been suggested that climate change could reduce the number of species globally (Thomas *et al*., 2004), although species richness might either increase or decrease at more regional scales (Menendez *et al*., 2006). Empirical observations have already indicated shifts in the distributional ranges of many species; these shifts are often consistent with 'global warming' as a driving mechanism (Parmesan and Yohe, 2003). In the North Atlantic, the link between large-scale patterns of pelagic diversity and increased ocean temperature has led to northwards expansions of species bringing about local biodiversity changes (Beaugrand *et al*., 2002; Hiddink & Hofstede, 2008). Predictive modeling of species' distributions under climate change scenarios predict further displacements for marine mammals (Kaschner *et al*. 2011), fish (Lenoir *et al*. 2011) and benthic macrofauna (Rombouts *et al*. 2012) in European seas over the next decades, especially for species at the edge of their southern distribution. As a result, current biodiversity hotspots may not remain static over time and so potential changes in local diversity could be monitored in relation to changes in large-scale environmental conditions.

In terms of the types of data that may be used to inform on biodiversity hotspots, WGBIODIV considered that studies should give due consideration to issues of catchability (e.g. how effectively a gear may sample different habitats within a survey area), survey design (e.g. tidal and seasonal effects) and sampling protocols (e.g. the taxonomic resolution of data collection and the time spent processing complex catches or sorting small individuals). Some faunal surveys may sample recruitment pulses, which can skew abundance data, and such data may also influence our perception of biodiversity hotspots.

Particularly in the case of sampling involving towed gears, there is the possibility of individual hauls sampling multiple habitats if the study area is characterised by high habitat heterogeneity. Areas of high habitat heterogeneity or the zones between distinct assemblages (ecotones) may be diverse, but may not necessarily be representative or typical of the constituent biotopes. On a broader scale, it has been noted that some biodiversity hotspots can be found in areas of 'ecological transition' (Araujo, 2002).

Many studies on biodiversity hotspots have focused on specific habitat types, such as coral reefs and seamounts (e.g. Roberts *et al*., 2002; Morato *et al*., 2010), or regional studies of specific faunal groups, such as polychaetes (Canales-Aguirre *et al*., 2011), sponges (Hooper *et al*., 2002; Samaai, 2006), hydroids (Hobbs *et al*., 2009), macroalgae (Phillips, 2001), echinoderms (O'Loughlin *et al*., 2011), or fish (Allen, 2008; Contente, 2011). Many of these studies have been conducted in either tropical or southern ocean areas, and there have been fewer studies on biodiversity hotspots within the ICES area (but see Hiscock & Breckels, 2007; Danovaro *et al*., 2009; Vandepitte *et al*., 2010).

GIS modelling has advanced our knowledge of the spatial distribution of key species and habitats, which can be used to inform conservation management and spatial planning, and to map areas of importance to marine biodiversity. These methods, with the caveat that the reliability of the models used must be acknowledged and that groundtruthing is required to validate models, provide a good addition to the tools applicable for the implementation of the Marine Strategy Framework Directive (MSFD). Such models have been published for example for modelling the distribution of:

- Juvenile flatfish (Florin *et al*., 2009);
- Economically important fish species such as founder, salmon, snapper, whiting (Morris & Ball, 2006);
- Fish-habitat relationships, including fish biomass, species richness and diversity (Knudby *et al*., 2009);
- Habitat-forming macroalage species, e.g. *Laminaria hyperborea* and other kelp (Bekkby *et al*., 2002, 2008, 2009; Méléder *et al*., 2010; Leaper *et al*., 2011);
- Seal habitat selection (Bekkby *et al*., 2002);
- Seagrass (Bekkby *et al*., 2008);
- Benthos (in general), including broad-scale mapping of a variety of organism groups (Holmes *et al*., 2008);
- Natura 2000 habitat (Bekkby & Isaeus, 2008).

Model development is partly correlated with the general improvement of desktop computers capacity and on the range of available software and to some extent this development has been driven by the availability of suitable datasets of empirical data.

The use of models has also been applied to model some of the pressures on the marine biota, e.g. sensitivity to oil spills (Moe *et al.*, 2000). Recently spatial models have also been developed to visualise relationships between cumulative human pressures, sensitive marine landscapes and landscape vulnerability. Such frameworks make it possible to assess the consequences of potential marine planning objectives and describe map uncertainty-related changes in management measures (Stelzenmüller *et al.*, 2010).

4.3 Integration of biological data from disparate sources

Integrating biological data from disparate sources was reviewed briefly by WGBIODIV last year (ICES, 2011), and is so only summarised here.

Qualitative information for a comprehensive spectrum of the marine fauna within an area is available for certain sites, usually areas in coastal and inshore areas. Such studies are exemplified by the regional faunal lists that have been compiled for various marine biological stations around Europe, including Plymouth (Marine Biological Association, 1957), Isle of Man (Bruce *et al.*, 1963), Milford Haven (Crothers, 1966), St Andrews (Laverack and Blackler, 1974) and Cullercoats (Foster-Smith, 2000).

Surveys of more offshore areas have generally been gear-specific studies of particular faunal groups (e.g. fish), with some more dedicated multi-gear surveys of specific sites and/or habitats. Spatially comprehensive taxonomic information for many marine taxa are lacking for some taxonomic and/or functional groups.

Some modelling approaches (e.g. Ecopath) have estimated biomass per unit area for various regions (e.g. Araujo *et al.*, 2005; Lees & Mackinson, 2007; Mackinson & Daskalov, 2007), although such studies typically have a restricted number of taxa/faunal groups, and so may not accurately reflect some aspects of species diversity.

When compatible biodiversity information for a specific region is available for different taxonomic and/or functional groups, there is often a wish to combine this information to highlight the areas with the highest overall biodiversity. It is important to address the most appropriate taxonomic level, temporal and spatial scales that any data can be aggregated to. Derous *et al*. (2007b), as part of the Marbef project, introduced a concept of integrated biodiversity value maps that showed the relative value of subareas, based on judgment of biodiversity components, but this method requires a lot of system knowledge and is not based on species-specific metrics. However, it has been applied in different European locations, including Belgium (Derous *et al.*, 2007a), Netherlands (Forero, 2007), Azores (Rego, 2007), Scilly Archipelago (Vanden Eede, 2007), Poland (Weslawski *et al.*, 2009) and the Basque coast (Pascual *et al.*, 2011) 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 within a GIS and combining them into a single map.

As discussed in an earlier report (ICES, 2011), the questions are whether absolute data are used or that the data are transformed onto compatible ranking scales, and also whether different biodiversity indicators and their maps are weighted.

4.4 Case study: The effect of gear efficiency of trawl gears on the perception of spatial variation in demersal fish biodiversity

Fraser *et al*. (2008) mapped variation in demersal fish species richness and evenness derived from two groundfish surveys that used different trawl gears. The Q3 IBTS used a GOV trawl, while the Dutch BTS used an 8 m beam trawl. The two gears have quite different catchability coefficients (proportion of individuals in the path of the gear actually retained in the net) for different types of fish; the beam trawl generally has higher catchability coefficients for flatfish, and lower catchability coefficients for roundfish, compared with the GOV trawl (Fraser *et al*., 2007). The spatial distributions of one richness and two evenness indices derived from each gear were quite different (Figure 4.1). The perceived locations of hotspots of demersal fish biodiversity can therefore be profoundly influenced by the choice of fishing trawl used to sample the fish assemblage.

Fraser *et al*. (2007) developed catchability coefficients to correct estimates of density derived from the Q3 IBTS otter trawl catch data to estimates of "actual density on the seabed". These catchability coefficients were applied to the Q3 IBTS data and new species evenness maps generated based on the derived estimates of "actual density on the seabed". Updated density estimates to account for estimated catchability only affected species evenness. Species richness essentially uses presence/absence data, and multiplying zero density values by any reciprocal catchability coefficient raising factor still gives a zero product. These catchability-corrected maps differed substantially from the original maps based on the "raw" catch density data (Figure 4.2). Using these two sets of maps to identify demersal fish species evenness hotspots, which might for example be used to inform the selection of MPAs intended to safeguard areas of high fish biodiversity, generated quite different area selections with very little spatial overlap (Fraser *et al*. 2009).

Determining spatial variation in fish biodiversity is therefore highly dependent on the gear used in the survey and profoundly influenced by variation in the catchability of each species in the particular gear chosen. In each case, possible explanations for the generated diversity maps were plausible. Maps of species diversity that took account of catchability in the survey gear revealed demersal fish species evenness to be greatest in the shallow, hydro-dynamically mixed southern North Sea where primary productivity is greatest (Reid *et al*., 1990; Berx & Hughes, 2009). High productivity is widely recognized as a positive correlate of species diversity (Brown, 1975; Davidson, 1977; Adams & Woodward, 1989; Currie, 1991; Hawkins *et al*., 2003; Whittaker & Heegaard, 2003; Rex *et al*., 2005). Maps of species diversity that did not take account of catchability in the survey gear revealed demersal fish species evenness to be greatest in the middle of the North Sea where change in depth gradients were steepest. Water depth is an important aspect of the physical habitat influencing spatial variation in the abundance of many fish species (Greenstreet *et al*., 1997; Hinz *et al*., 2006). Higher habitat variability is also widely acknowledged as a positive correlate of species diversity (Power, 1972; Fox, 1983; Abbott and Black, 1987; Rosenzweig, 1995).

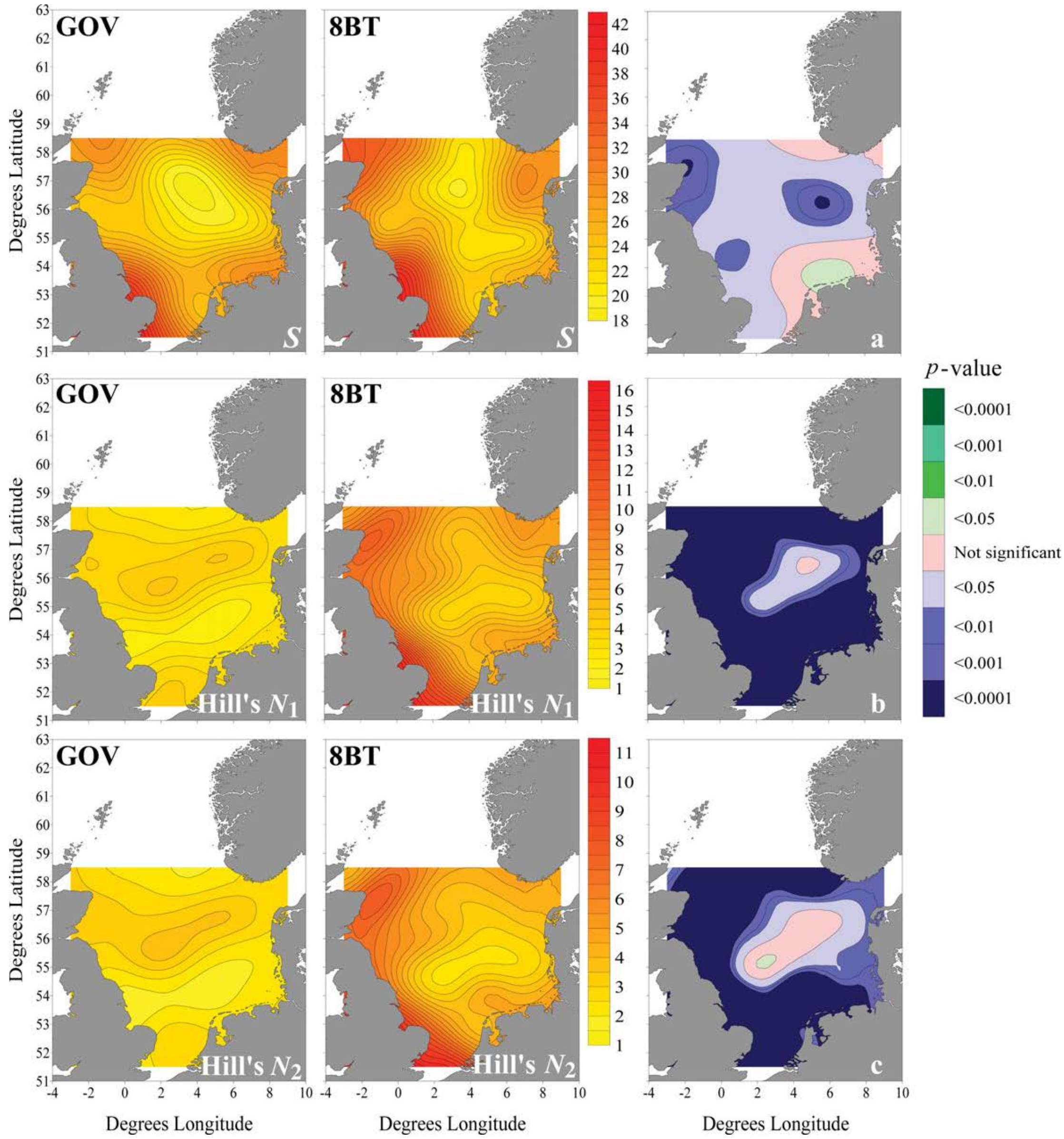

Figure 4.1. Spatial variation in three diversity metrics (S, species richness; N1, Hill's N1 index of species diversity and N2, Hill's N2 index of species diversity) computed for the IBTS (GOV) and Dutch BTS (8m BT) surveys. Differences between the three measures of species diversity are shown in panels (a), (b), and (c), where green/blue indicates that the metric for the GOV is significantly greater/less than that for the 8mBT (From Fraser *et al.*, 2008).

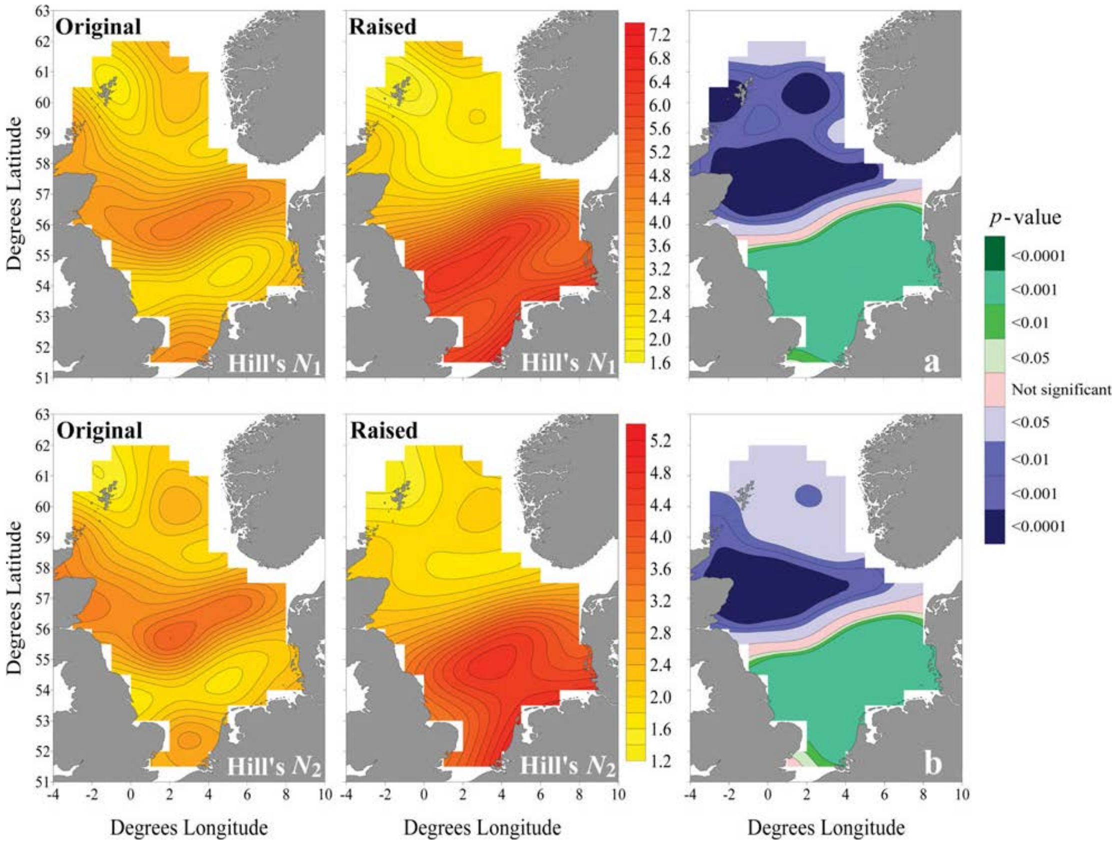

Figure 4.2. Spatial variation in Hill's N1 and Hill's N2 across the North Sea based on the IBTS (GOV) dataset, illustrating the effect of taking into account species- and size-related estimates of catchability in the GOV. Differences between the two measures of diversity are shown in panels (a) and (b), where green/blue indicates that the metric for the original data is greater/less than that for the raised GOV data (From Fraser *et al.*, 2008).

4.5 Case study: Modelling fish distributions as a tool for predicting species abundance hotspots

Although fish distributions may vary in space through time, areas where particularly high densities are observed may remain fairly constant in both space and time (Hinz *et al.*, 2003). This has been related to the concept of "essential fish habitat" (EFH) (Benaka, 1999); the idea that different species have particular habitat requirements and therefore show particular preference for and specificity to locations where these habitat features are present (Hinz *et al.*, 2006). This leads to the idea that fish distributions might be modelled on the basis of their habitat preferences, and that such models might therefore be useful in predicting species abundance "hotspots".

Detailed knowledge of seabed habitats is difficult and time consuming to obtain by direct sampling of the seabed using grabbing or dredging approaches. However, a variety of acoustic remote sampling techniques are now available that present opportunities to map large areas of seabed rapidly whilst the vessel is underway (Brown *et al.*, 2005). Detailed seabed habitat maps at relatively high spatial resolution have been developed using such methods and shown to convey information that is useful in interpreting fish distributions (Greenstreet *et al.*, 1997; Greenstreet *et al.*, 2010). Here we review the results of a study examining the utility of using the acoustic data gathered during remote seabed mapping surveys to model fish distributions directly. If successful, this would suggest that remotely collected seabed acoustic data might be used directly to predict fish distributions and identify potential fish abundance hotspots without the need for ground-truthing seabed sampling that is usually required for the intermediate step of generating seabed habitat maps.

RoxAnn seabed acoustic characteristic and water depth data, collected whilst the vessel was engaged in pelagic fish acoustic and seabird-at-sea survey work in a 4500 km^2 study area immediately east of the Firths of Tay and Forth in southeast Scotland (Greenstreet *et al*., 2006; Daunt *et al*., 2008), were kriged to derive interpolated maps of E1 (seabed roughness), E2, (seabed hardness) and water depth at a 1 km^2 pixel resolution (Greenstreet *et al*., 2010). Demersal fish populations in the study area were sampled at 19 trawl stations (Greenstreet *et al*., 2006) between mid May to mid July in each year from 1999 to 2008 except 2004, giving nine replicates at each station (a total sample size of 171 trawl samples). During each 30 min. trawl operation, the position of the vessel was recorded at 5 min. intervals, giving a total of 7 positions per trawl. Overlaying these trawl position data across the 1 km^2 pixel resolution E1, E2 and depth layers using IDRISI GIS software allowed the acoustic seabed characteristics to be determined at each position. This enabled each demersal trawl sample to be characterised in terms of the acoustic seabed characteristics present along its seabed footprint. Eight demersal fish species, each constituting at least 1%, and between them making up 98%, of the total of 166 624 fish sampled were therefore deemed sufficiently abundant as to model their distributions (Table 4.1). Examination of the length frequency distributions revealed juvenile components in the samples of each of these species. Adult and juvenile fish often have different spatial distributions. Here only the adult components were modelled. Table 4.1 indicates the length threshold that fish of each species had to exceed in order to be included in the analyses.

Species abundance data were standardised to number per 30 000 m^2 and generalised linear models of the form $X_s = \alpha D + \beta E_1 + \delta E_2 + C$ were derived where *D* is depth, E_1 and E_2 are RoxAnn E1 and E2 respectively and *C* is a constant. Table 4.2 shows parameter values for only the most significant terms giving parsimonious models. For plaice, none of the parameters were significant and for whiting and Norway pout the final models were not significant at the $P<0.05$ level. Highly significant models (*P* ranging from <0.001 to 0.009) were derived for the remaining five species (common dab, long rough dab, lemon sole, haddock and grey gurnard), and these models explained between 51% and 65% of the observed spatial variation in abundance.

A jacknife procedure was used to validate the models for these five species. For each species, data collected in each year in turn was excluded from the model parameterisation process and parameter values were determined just from the data collected during the remaining eight years (Table 4.3). The parameter values obtained were substituted into the model and used to predict the distribution in the excluded year. The expected values predicted by the models were compared with actual observed data at each station in each year. Predictive power was high, explaining between 39% and 62% of variance in the observed numbers of fish per 30 000 m^2 swept area.

This example demonstrates that it is possible to develop simple models that are capable of using the acoustic data collected during remote seabed surveys directly to predict the distributions of those demersal fish species that have a close affinity to the seabed. Adult whiting and Norway pout spend a good fraction of their time feeding on prey species that are primarily encountered in the water column rather than on the seabed (Albert, 1994; Hislop *et al*., 1991; Greenstreet *et al*., 1998), so perhaps it was not surprising that the models for these two species were not statistically significant. Plaice, however, as a flatfish, might have been expected to have a close affinity to the seabed, and it was perhaps surprising therefore that none of the acoustic seabed characteristic variable seemed to influence plaice distribution. Plaice are amongst the most abundant of flatfish in the North Sea and are one of the most widely distributed; they may simply be able to use almost all benthic habitats relatively efficiently and not have any special preference for, or specificity to, any particular habitat type. In an earlier study, plaice only demonstrated any preference for particular sediment habitats in water shallower than 35 m (Greenstreet *et al*., 1997). All but one of the trawl stations in this study were located in water deeper than 35 m.

These models may have the capacity to identify species density hotspots at a much finer spatial resolution than might be achieved directly from trawl survey data, conditional on the model, and spatial/temporal scale of the data. Such approaches may not always be extrapolated to other areas, and applying these approaches should involve ground-truthing the predictions. Nevertheless, such information could be especially useful in defining relatively small areas for marine spatial planning purposes. The models also integrate all the available information in an objective way, and thereby help to address some of the problems associated with sampling variability that might otherwise occur if trawl sample data are used directly to define density hotspots.

Table 4.1. The number of fish of each species sampled, the percentage of the total sample, and the length threshold that individuals of each species had to exceed in order to be included in the analysis.

Species	Number sampled	Percentage of total	Length threshold
Common dab *Limanda limanda*	55 369	33	6 cm
Long rough dab *Hippoglossoides platessoides*	9655	6	6 cm
Plaice *Pleuronectes platessa*	6067	4	6 cm
Lemon sole *Microstomus kitt*	1759	1	6 cm
Haddock *Melanogrammus aeglefinus*	38 896	23	13 cm
Whiting *Merlangius merlangus*	46 966	28	9 cm
Norway pout *Trisopterus esmarki*	3168	1	10 cm
Grey gurnard *Eutrigla gurnardus*	1562	1	13 cm

Table 4.2. Results of generalised linear modelling showing significant parameter values and overall goodness of fit (R^2) and significance (P) of the overall model.

Species	α	β	δ	*C*	R^2	*P*
Common dab	-0.060	-6.561	3.583	2.718	0.528	0.009
Long rough dab	0.070		-2.006	-1.021	0.649	0.000
Plaice					0.000	1.000
Lemon sole	0.060		1.969	-5.410	0.589	0.001
Haddock	0.071			-3.536	0.616	0.000
Whiting			-0.997	1.221	0.123	0.140
Norway pout	0.038			-1.881	0.189	0.063
Grey gurnard		-3.851	2.643	-0.830	0.508	0.003

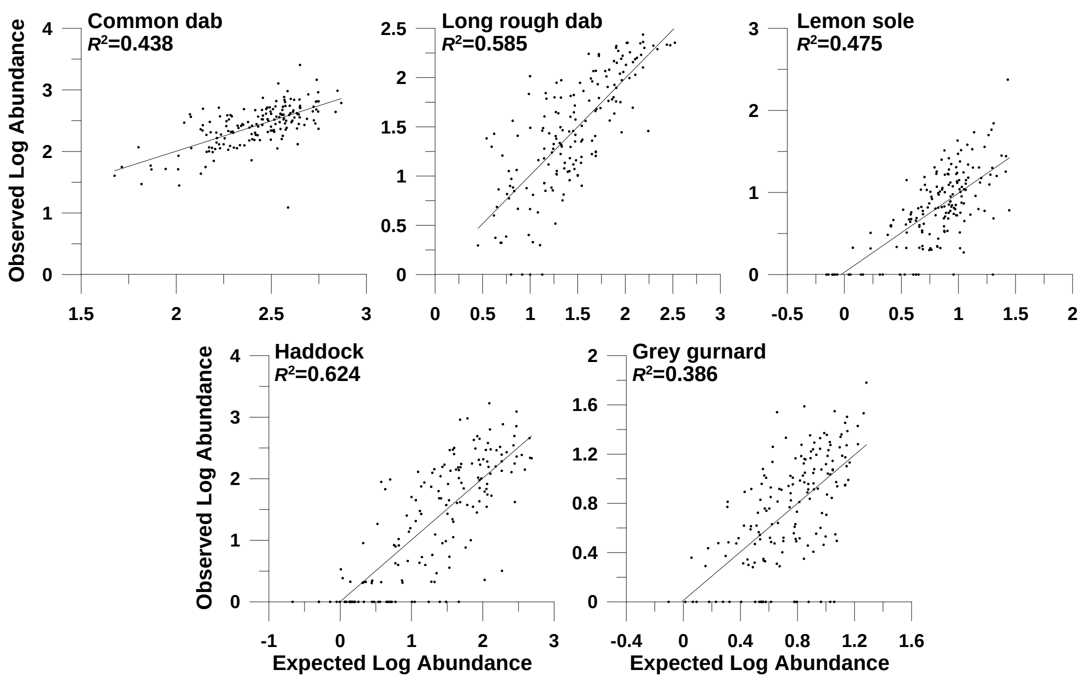

Figure 4.3. The jacknife results showing a comparison of observed log-numbers with expected log-numbers of common dab, long rough dab, lemon sole, haddock, and grey gurnards per standard 30 000 m² swept-area tow at each of 19 demersal stations in 9 years (n = 19 x 9 = 171) predicted by the habitat-choice models.

Table 4.3. Individual parameters used for the models for each species in each year in the jacknife validation procedure.

Species	Year	*C*	*α*	*β*	*δ*	R^2	*P*
Common dab	1999	2.699	-0.062	-7.020	3.905	0.590	0.003
	2000	3.158	-0.064	-6.900	3.558	0.494	0.015
	2001	3.417	-0.070	-7.311	3.808	0.509	0.012
	2002	2.448	-0.054	-6.258	3.405	0.488	0.016
	2003	2.835	-0.063	-6.551	3.603	0.561	0.005
	2005	2.574	-0.059	-6.296	3.522	0.529	0.009
	2006	2.141	-0.052	-5.832	3.336	0.480	0.018
	2007	2.778	-0.060	-6.664	3.594	0.537	0.008
	2008	2.410	-0.057	-6.216	3.519	0.527	0.009
Long-rough dab	1999	-1.008	0.068	0	-1.954	0.619	0.000
	2000	-4.335	0.105	4.776	-3.162	0.695	0.000
	2001	-0.731	0.066	0	-2.109	0.630	0.000
	2002	-1.106	0.072	0	-2.011	0.533	0.000
	2003	-4.596	0.109	4.979	-3.233	0.704	0.000
	2005	-1.315	0.073	0	-1.914	0.659	0.000
	2006	-1.123	0.070	0	-1.952	0.652	0.000
	2007	-1.089	0.070	0	-1.962	0.643	0.000
	2008	-4.857	0.111	4.956	-3.083	0.689	0.000
Lemon sole	1999	-5.523	0.063	0	1.946	0.602	0.001
	2000	-5.321	0.059	0	1.952	0.581	0.001
	2001	-5.518	0.060	0	2.057	0.605	0.001
	2002	-5.328	0.056	0	2.059	0.581	0.001
	2003	-5.334	0.058	0	2.009	0.564	0.001
	2005	-5.640	0.064	0	1.992	0.603	0.001
	2006	-5.398	0.061	0	1.920	0.591	0.001
	2007	-9.961	0.114	6.853	0	0.575	0.001
	2008	-10.074	0.116	6.866	0	0.598	0.001
Haddock	1999	-3.744	0.075	0	0	0.595	0.000
	2000	-3.508	0.070	0	0	0.580	0.000
	2001	-3.453	0.069	0	0	0.576	0.000
	2002	-3.459	0.069	0	0	0.591	0.000
	2003	-5.940	0.090	2.343	0	0.692	0.000
	2005	-6.281	0.093	2.622	0	0.660	0.000
	2006	-3.466	0.069	0	0	0.613	0.000
	2007	-3.462	0.069	0	0	0.624	0.000
	2008	-3.564	0.071	0	0	0.642	0.000
Grey gurnard	1999	-0.624	0	-4.094	2.599	0.495	0.004
	2000	-0.564	0	-4.369	2.690	0.545	0.002
	2001	-1.013	0	-3.838	2.786	0.524	0.003
	2002	-1.154	0	-3.671	2.816	0.563	0.001
	2003	-0.957	0	-3.914	2.779	0.553	0.002
	2005	-3.975	0.033	0	1.898	0.502	0.004
	2006	-0.524	0	-4.269	2.607	0.507	0.004
	2007	-0.727	0	-3.550	2.406	0.419	0.013
	2008	-0.794	0	-3.473	2.421	0.418	0.013

4.6 Case study: Identifying hotspots in the Barents Sea

In the Barents Sea, monitoring is a joint effort between Norway and Russia, based on collaboration between the two countries since the 1950s. In an attempt to improve the efficiency and to enhance the ecological and scientific merit, a joint Norwegian acoustic survey (including CTD measurements for salinity), an 0-group survey using pelagic trawl and several demersal surveys were gradually merged to form the Barents Sea ecosystem survey in 2003. The ecosystem survey sampling protocol also emphasises good taxonomic resolution and the identification of benthos and fish in catches. The same methods for acoustics, pelagic trawl and demersal trawling (since 2004) are applied on board both Norwegian and Russian vessels, although the methods for zooplankton sampling and seabird and sea mammal observations have differed. These data could be used to identify hotspots across taxa and trophic levels (Anon, 2011, Figure 4.4). The Barents Sea is a transition zone between Atlantic and Arctic water masses, and for Arctic and boreal fauna and flora. This is partly detectable in the maps made from data collected at the ecosystem survey (Figure 4.4).

Figure 4.4.a shows the distribution of different taxonomic groups of benthos caught as by-catch in demersal trawls. There were large concentrations of sponges in the south-western entrance of the Barents Sea. The central Barents Sea was dominated by echinoderms, as was the north eastern area. In the south east, crustaceans dominated.

The demersal fish species composition differed between the northern Arctic part and the southern area that is dominated by the inflow of Atlantic water (Figure 4.4.b). There were three coastal/shallow water areas differing in species composition. The one in the south west partly coincided with the area with high concentrations of sponges and was the area with the highest density of demersal fish species (Figure 4.4.b).

The acoustic data on the main pelagic species in the Barents Sea showed that they have segregated spatial niches: juvenile herring were found in Atlantic water masses, capelin was found in the frontal area between Arctic and Atlantic waters, and polar cod was found in Arctic water (Figure 4.4.c).

0-group fish were sampled with pelagic trawls. Of the most abundant species, herring, haddock and cod, all spawned outside the Barents Sea, are the 0-groups spread into the Barents Sea with the Atlantic inflow and distributed over large areas (e.g. cod Figure 4.4.d).

Different sea bird and sea mammal species from northern and southern assemblages (Figure 4.4.e–i). There were particularly many sea bird observations around the Svalbard archipelago, where there are large breeding colonies. It should be noted that sea bird and sea mammal observations were dependent on favourable weather conditions.

The zooplankton biomass data were sampled by plankton nets that mainly sampled mezo-zooplankton, and were dominated by copepods. The spatial patterns reflected the Atlantic inflow, local production and local depletion by pelagic fish (Figure 4.4.j).

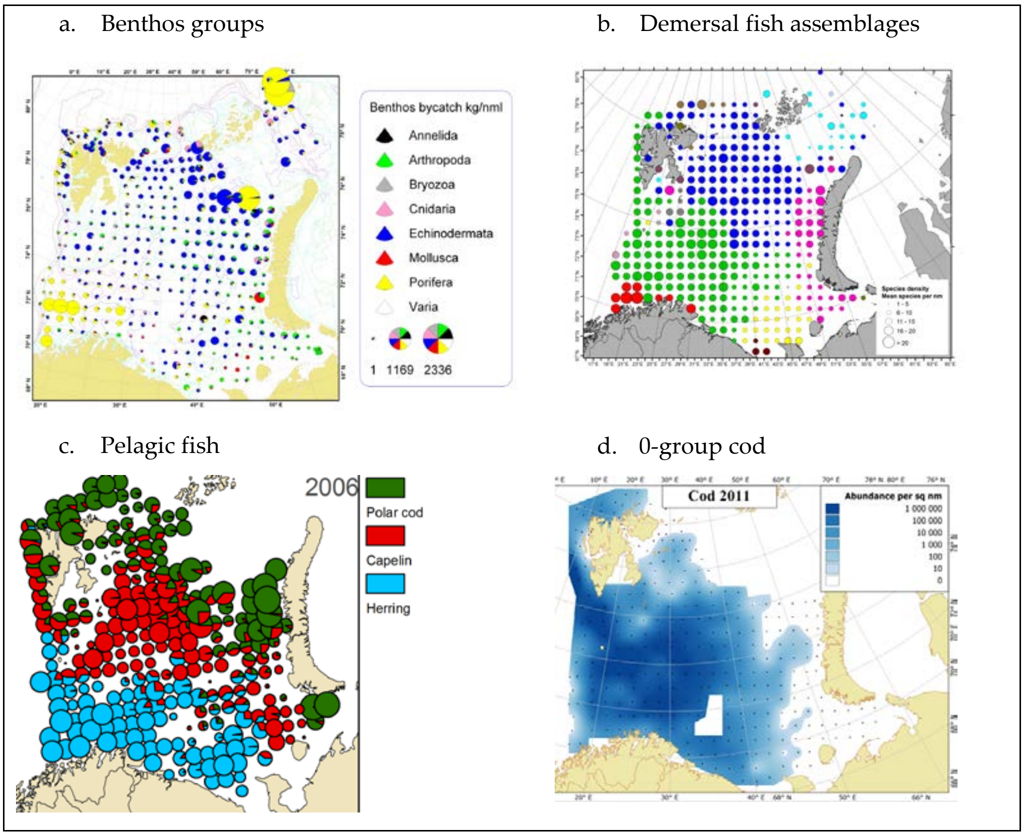

Figure 4.4. Barents Sea ecosystem survey showing (a) the relative distribution of the main benthic animal groups (presented as quantitative circles at each sample station) as sampled by Campelen trawl (August–October 2011); (b) demersal fish assemblages identified from cluster analysis on presence-absence data (gridded) from Campelen trawl survey data (2004–2009). The size of the circles is proportional to the average number of species caught per trawl station by grid station. Circles with same colour have similar species compositions (from Johannesen *et al.*, submitted); (c) acoustic registration of capelin, herring and polar cod during the 2006 ecosystem survey; (d) distribution of 0-group cod (data from August–October 2011 and interpolated from trawl catches).

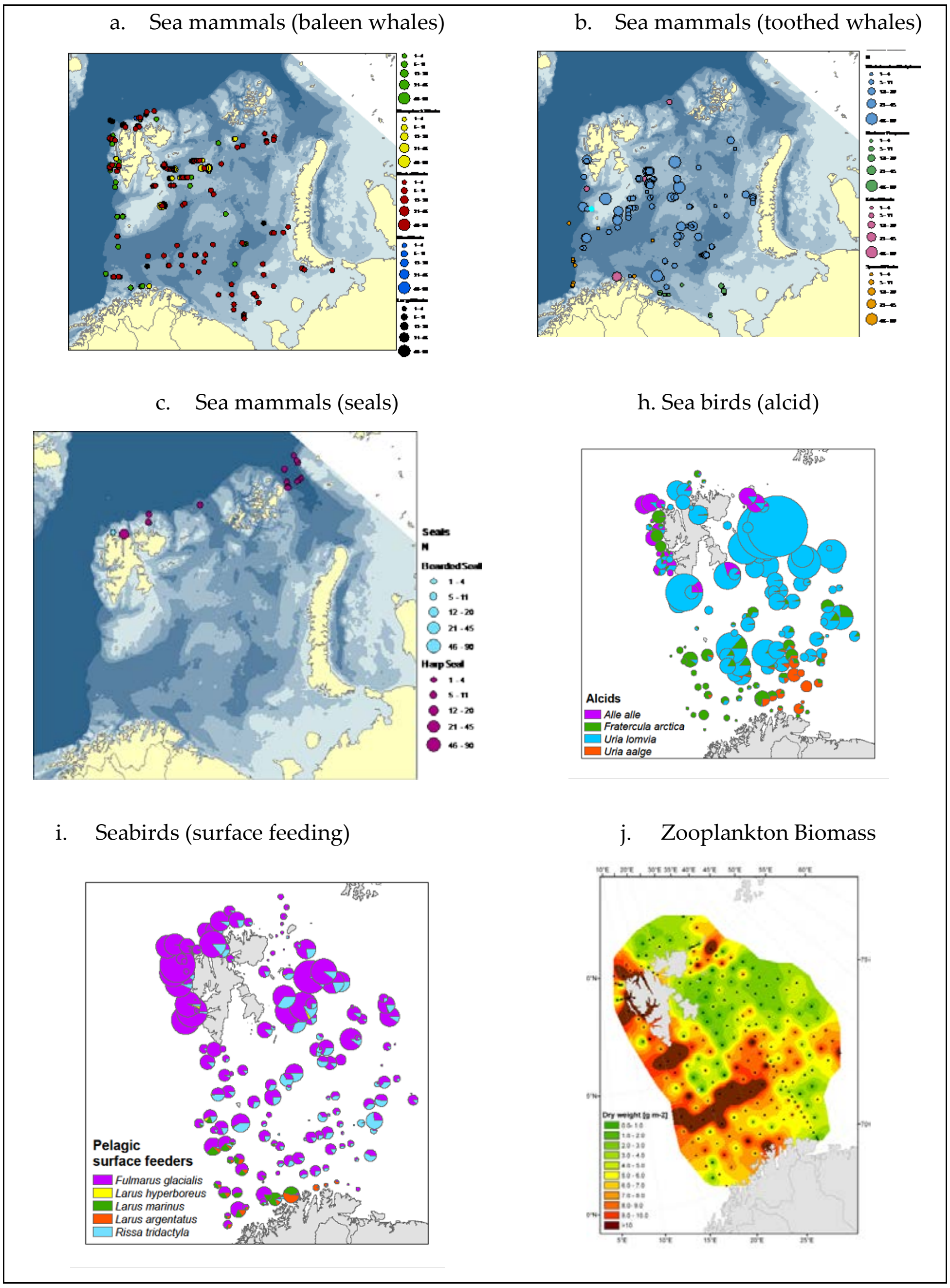

Figure 4.4 (continued). Barents Sea ecosystem survey showing the distribution of (e) baleen whales, (f) toothed whales and (g) seals, (h) alcid seabirds, and (i) surface-feeding seabirds, all as observed during the ecosystem survey (August–September 2011); and (j) zooplankton biomass from wp2 nets. Most of these data were collected on the Joint Russian-Norwegian ecosystem survey and the figures adapted from Anon (2011).

4.7 Case study: Mapping biodiversity in Dutch waters

Bos *et al*. (2011) examined biodiversity hotspots in Dutch waters from a MSFD perspective and found that the results for different species groups such as benthos, fish, birds and marine mammals could best be presented separately, instead of being combined in a final map.

To be able to compare and combine maps, they standardized them by rescaling the underlying values to a standard scale of 1–5 (low to high values). The aim was to combine different maps into a single map, by adding the maps and rescaling the obtained values again on a scale from 1–5. In order to be able to aggregate data at different spatial scales, they used a grid of 5x5 km as the basis. Mapping of biodiversity hotspots on the level of individual biodiversity metrics within taxonomic groups proved to be useful. Separate maps of biodiversity metrics were therefore most informative. There are many different way in which such maps can be aggregated and the method used and the way in which data are split into classes will influence the maps.

Spatial patterns of benthic biodiversity were more consistent than for other taxonomic groups. This was probably due to their sedentary lifestyle. For fish, spatial biodiversity patterns were less clear than for benthos, probably because fish are more mobile. Although birds are mobile species as well, some areas had consistently higher bird values than others. In the coastal zone this was caused by the higher number of species present (coastal birds). For marine mammals, biodiversity patterns were difficult to interpret. This is partly due to the data constraints and the low number of species. For this group, it is probably best to consider the species separately, as opposed to the taxonomic aggregation applied for benthos, fish and birds.

4.8 Case study: HELCOM Work related to biodiversity hotspots

In 2003, at the Joint HELCOM-OSPAR Ministerial Meeting, the Contracting Parties of these two Conventions agreed on a Joint Work Programme. One of the aims was to complete, with the Birds and Habitat Directive's Natura 2000 sites, a well managed and ecologically coherent network of Baltic Sea Protected Areas (BSPAs) and OSPAR MPAs, by 2010. This goal was later reaffirmed and expanded in the HELCOM Baltic Sea Action Plan (HELCOM, 2007) by explicitly requesting that an assessment of the ecological coherence of the BSPA network together with the marine Natura 2000 and Emeral sites should be done (HELCOM, 2007; Boedecker, *et al*. 2010). The main issue was to address the possible redundancies, protection deficiencies and biogeographical gaps created by the *ad hoc* approach in selecting MPAs up until present (Boedecker *et al*., 2010).

The work carried out to fulfil these commitments built on the experiences and results from the BALANCE project (Interreg IIIb 2005–2007), in particular the use of the site selection software (and decision support tool) MARXAN, and the MARXAN interface Zonae Cognito. These were used to identify a set of achievable representative networks of MPAs. MARXAN can produce suitable planning areas that accomplish a number of ecological social and economic objectives. MARXAN aims to achieve the optimal user-defined biodiversity targets in the most cost-efficient way with minimum "cost" that can be a monetary value or variable related to the objectives as defined by the user (HELCOM 2010).

MARXAN made it possible to address issues of adequacy (i.e. that MPAs should meet certain size criteria, indicator species, and habitats/biotopes), replication (selected species and biotopes should be well represented in the network), representativity (selected species, biotopes, marine landscapes (broad-scale habitats) and inshore/offshore representation criteria should be met), and connectivity (by acknowledging a theoretical and species specific dispersal distances of 25 km and 50 km respectively).

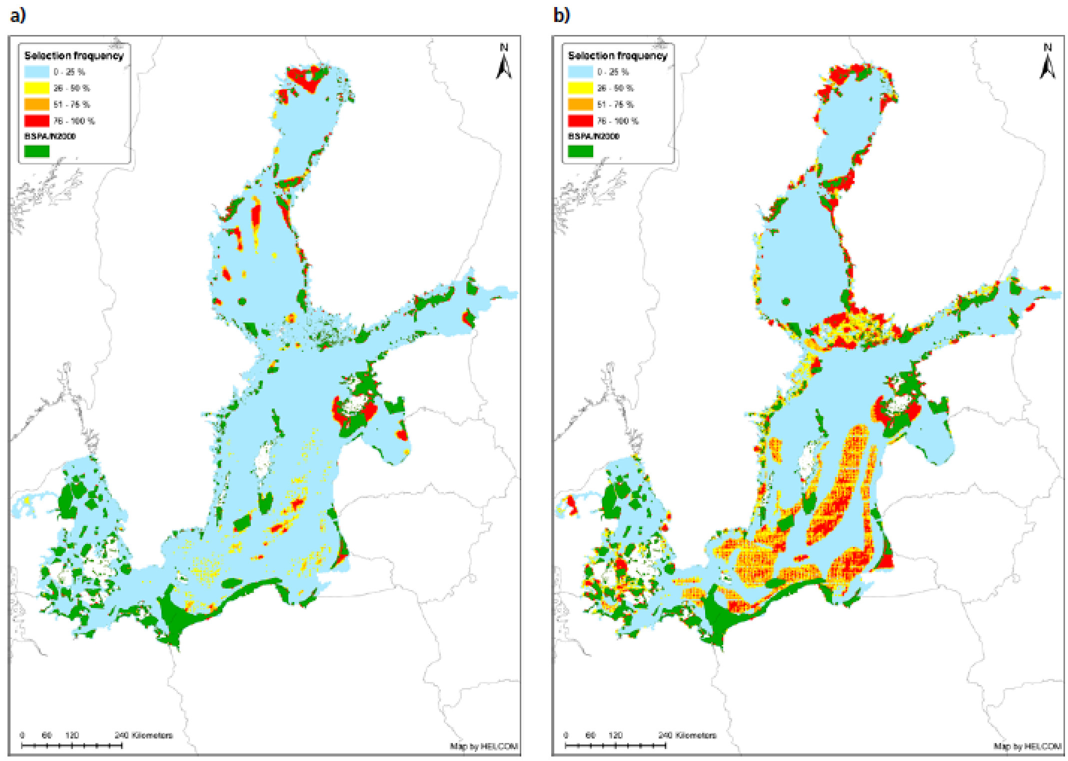

Figure 4.5. Maps showing two alternative results from the MARXAN analysis, based on (a) lower, and (b) higher conservation targets with a minimum 12% sub-regional coverage. Existing BSPAs and marine Natura 2000 sites included and with a cod spawning area data layer. Source: HELCOM (2010).

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5 New technologies and approaches to support improved biodiversity monitoring and assessment

5.1 Introduction

Despite the various new technologies that can support biodiversity studies, biodiversity monitoring has, through its taxonomic component, a strong conservative aspect that needs to be retained when trying to achieve reliable and comparable results. This comprises not only keeping and retaining trained personnel and expertise, but also to support ongoing training workshops and Quality Assurance routines in all identification programmes.

New technologies, such as high definition television, marine acoustics, marine electronics, new possibilities in SPI technology etc. will clearly augment existing monitoring programmes (Smith & Rumohr, 2012). New technologies in laboratory work, such a genetic bar coding and other genetic tools will facilitate improved taxonomic discrimination. Great potential lies in the use of computer-aided identification of species that is also aided effectively by web-based identification and taxonomic repositories.

The way to a computerized "Coulter Counter" for the identification of benthic invertebrates and fish may nevertheless be still quite long.

However, for long-term biodiversity monitoring, there is a clear and fundamental need that existing long-term monitoring programmes (e.g. the IBTS and CPR surveys) are retained for standardised data collection. Supplementary field methods and special technologies, as and when incorporated into monitoring programmes, should typically be additional studies, and not necessarily result in changes to survey design.

5.2 New technologies for the monitoring and assessment of various aspects of biodiversity

5.2.1 Acoustic seafloor habitat mapping

According to Smith & Rumohr (2012), acoustic mapping techniques are an essential part of the imaging approach in recording physical attributes, habitat and community patterns of seafloor habitats at different spatial scales. A handbook for seafloor imagery analysis was also produced by Blondel & Murton (1997). Acoustic devices may be ranked according to their resolution and area of coverage. Historically single beam echo-sounders developed for depth measurement have been used to depict bottom structure as well as some sedimentological properties depending on the reflecting properties of the sea-floor. For all acoustic methods the basic principles apply that the lower the frequency the longer the range, but the higher the frequency, the greater the resolution.

Single beam echo-sounders, originally developed for depth measurements, have been used to depict bottom structure as well as some sedimentological properties, depending on the reflecting properties of the sea-floor. Single beam echo-sounders (30 kHz – 3.5 MHz) and acoustic ground discriminating systems (AGDS) allow a variety of information about the reflective characteristics of the seafloor to be collected. Such systems highlight the contours and depths of the seabed, and indicate the thickness and structure of sediment layers. Recent developments apply the analysis of echo-sounder returns to include multibeam systems therefore enabling wider swathes to be analysed.

Side-scan sonar is an acoustic imaging device (100–1000 kHz) used to provide wide-area, high resolution pictures of the seabed. Originally developed for marine geology, it is now used routinely in benthic ecology. This method allows seabed features (e.g. reefs, sand ripples, seagrass beds) to be mapped and can reveal some distinct sediment structures, whether of biogenic (e.g. feeding mounds, feeding depressions) or anthropogenic origin (e.g. wrecks, trawl tracks).

Modern high frequency side-scan sonar devices provide a high resolution image of the seabed, and may detect objects in the order of tens of centimetres at a range of up to some 100 metres on either side of the tow fish. Side-scan sonar can produce, under optimal conditions, an almost photo-realistic picture of the seafloor. Over several geo-referenced swathes, a mosaic image can be built up forming an area map, where geological, sedimentological and some biological features are discernible. Considerable increases in resolving capacity down to centimetres may be available in the future from the full use of synthetic aperture sonar (SAS) in seafloor imaging.

Side-scan sonar may play a larger role in the future in seabed discrimination, and side-scan processing software is becoming available that will automatically classify features on the seabed. As in single beam AGDS, ground-truthing (e.g. sediment grabs, underwater photographs) is still necessary with higher resolution methods.

Swathe bathymetry through hull-mounted multibeam or inferometric methodologies, is a relatively newer seabed mapping technology that produces high density geo-located depth measurements through digital processing techniques, and can be used to create impressive shaded-relief or colour topographic maps. A major advantage of multi-beam systems over side-scan sonar is that they generate quantitative bathymetric data which may be used for habitat classification. They may also utilise backscatter data to form images similar to those of side-scan data, albeit with lower resolution, partly due to the variable height above the seabed of the hull-mounted sensors. The beam width makes them less useful for object detection when the objects are less than 1 m^2 and they require accurate information on navigation, roll, pitch and sway and the calibration of sound velocity.

Acoustic mapping techniques are an essential part of the imaging approach in recording physical attributes, community patterns of seafloor habitats at different spatial scales. In terms of biodiversity monitoring, such techniques and advances in data analysis will aid with studies of benthic habitats and some associated species (where characteristic features are visible), and facilitate more detailed mapping, as described below.

Mapping

The use of acoustic information has facilitated broadscale mapping. There is an implicit requirement for continuous mapping that can be applied across regions due to the Marine Strategy Framework Directive (MSFD). One of the initiatives that respond to this requirement is the EU SeaMap project. It is a "Preparatory Action" for development and assessment of a European broad-scale seabed habitat map (EC contract no. MARE/2008/07, funded by the EU Commission, DG MARE). The main goal of the project is to compile broad-scale habitat maps for a large part of Europe's seas (Figure 5.1).

The project has harmonised and improved methods used to produce the MESH EUNIS seabed habitat maps for the North Sea and Celtic Seas, merging these with the seabed maps of the Baltic Sea from the BALANCE project, and extending the methodology to the western basin of the Mediterranean Sea. Through expert application of the EUNIS classification and improved input data layers and seabed habitat modelling techniques, existing maps were refined, and their coverage extended to the specified Marine Regions, and the seabed habitat maps cover nearly 2 million km^2. The EU SeaMap has also created confidence maps associated with the seabed habitat maps in order to visualize the variation in quality and resolution of the input data layers (Cameron and Askew, 2011).

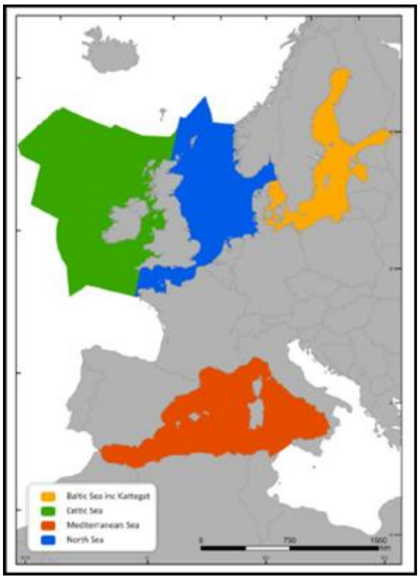

Figure 5.1. The geographic extent of the EU SeaMap project. Map from the EU SeaMap final report (Source: Cameron and Askew, 2011).

The EU SeaMap has compared maps of the marine seabed habitats of Europe at an international level and overcome the difficulties in comparing across regions; from the differences in methods/classifications used; variations in scale etc. (Figure 5.2). This modelled seabed habitat map is freely available and an interactive web mapping portal (a web-GIS) allows users to view and query the data. Another route through which the data are made available is Web Mapping Services, which provides a 'live link' to the data, either directly to a desktop GIS, or to another internet server that hosts a different web-GIS, such as the European Atlas of the Seas or the other EMODnet portals. The EUSeaMap web-GIS uses the open source software MapServer and the OpenLayers API.

Although the EU SeaMap can provide data that can be used to inform on the MSFD and for Marine Spatial Planning, it is important to acknowledge that further studies (e.g. ground-truthing the habitat classes) may still be required for more detailed, applied analyses.

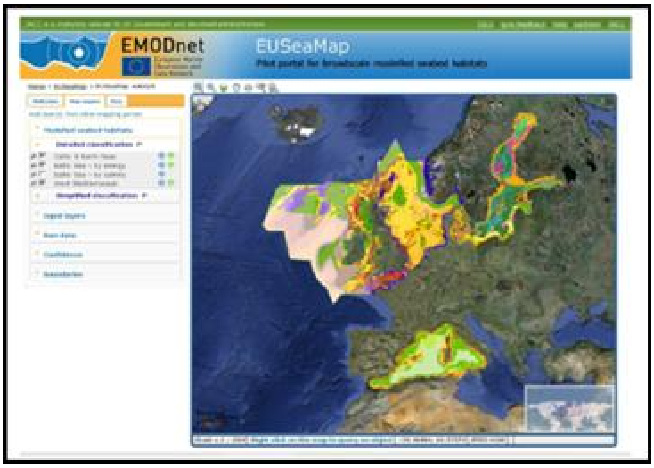

Figure 5.2. EUSeaMap project, highlighting the distribution of modelled seabed habitat maps in European waters. Source: http://ec.europa.eu/maritimeaffairs/emodnet/ preparatory/home.html

5.2.2 Sediment profile imagery (SPI)

Sediment profile imagery (SPI) or REMOTS (Remote Ecological Monitoring Of The Seafloor) utilises an imaging device in an inverted periscope (optical prism) that penetrates the sediment and facilitates the imaging of the sediment water interface and upper sedimentary layers (approximately 15 x 20 cm in area), allowing fine scale analysis of physical, chemical and biological features (Rhoads & Germano, 1982). Direct measurements can be made from the images and Rhoads & Germano (1982) developed a computer-aided system for analysis of such parameters as grain size, surface boundary roughness, mean (apparent) redox depth, methane gas pockets, thickness of over-lying (dredged/disposed) material, visible epifauna, tube density and type, faecal pellet layer, microbial aggregations, feeding voids, and dominant fauna. This development has made the collection of a variety of abiotic and biotic measures easier and quicker than with earlier methods (i.e. core sampling).

One of its primary uses is in large area surveys where local hotspots need to be identified and sampled in more detail. Systems are frame-mounted and wire-deployed for remote use or can be diver-operated for directed sampling in shallow waters and under fish-farms where boats cannot operate. Germano *et al*. (2011) comprehensively reviewed the various aspects of SPI technology, which has been increasingly used in the ICES area, particularly to better investigate bioturbation, the vertical distribution of benthic fauna within surface sediments and in helping relate infaunal communities with redox state (e.g. Birchenough *et al*., 2012).

Such technologies can help interpret the structure and function of benthic diversity, in relation to MSFD Descriptors 1, as well as sea-floor integrity.

5.2.3 Digital imagery and underwater photography

Increasingly, knowledge of broad-scale distribution patterns of populations, communities, and habitats of the seafloor is needed for impact assessment, conservation, and studies of ecological patterns and processes (Hewitt *et al*., 2004). In the last two decades the use of underwater imagery (video and still camera) techniques has become increasingly common in marine ecological research (Coggan *et al*., 2008). The image quality of cameras suitable for underwater use have improved greatly, now providing high resolution colour images at an affordable cost. These cameras can be used from a wide variety of stationary or mobile platforms such as:

- Divers (hand-held cameras);
- Stationary underwater support (tripods or other stationary platforms, e.g. for bait studies);
- Drop-video/still cameras;
- Remotely operated vehicles (ROVs);
- Autonomous underwater vehicles (AOVs), i.e. submersibles operating without an attached tethering cable for supplying power (Desa *et al*., 2006).

The more advanced platforms (ROVs or AOVs) may come with other sensors (e.g. acoustic devices or CTD sensors) attached.

Underwater imagery devices all share the ability to collect information (observe) faster and longer than what a diver would be able to, or collect information from depths or hostile environments where no diver would go to (Carbines & Cole, 2009). Underwater imagery is also used to ground-truth remote sensing data from acoustic devices or aerial photography (Hewitt *et al*., 2004, Lefebvre *et al*., 2009). Underwater imagery techniques are most successful in capturing macroscopic sessile species but, combined with other sensors or when using bait, images of more mobile species may also be sampled. By using 3D camera techniques (using two cameras) length measurements can be made of mobile species; alternatively lasers or autofocus camera systems can be used for the same purpose (Rochet *et al*., 2006).

Baited Remote Underwater Video Systems (BRUVS) have been widely used in some tropical areas to inform on the presence of some conspicuous piscivorous/scavenging species, and other forms of baited camera system have also been used in various parts of the ICES area, e.g. for deep-water fish species (Bailey *et al*., 2007). As with other visual studies, baited camera systems have limitations, in terms of the species sampled, accuracy of identification for smaller or more cryptic species, and that the data may not allow for rigorous statistical analyses of abundance. Nevertheless, they may have a role for informing on the functional role of some scavengers (e.g. fish, crustaceans, cirolanid isopods), and may allow for non-destructive sampling of some species where destructive sampling is not deemed appropriate (e.g. for fish of high conservation interest; for better understanding the fauna associated with biogenic habitats).

Limits to the applicability of these techniques for collecting information on the marine biodiversity must be highlighted. Potential sources of error must be known in order to make it possible to decide which conclusions can be drawn based on the collected data and when to be cautious. Examples of errors that may occur when applying underwater imagery techniques include:

- The temporal resolution is insufficient to capture all relevant events;
- Fast moving species may not be recorded (False absence);
- Failing to cover the entire annual succession of sessile species (False absence);
- Small/cryptic species are not recorded or are overlooked (False absence);
- The image quality is too poor to detect some macroscopic species (False absence);
- Mobile species are affected by photographic equipment (attracted or avoiding) and thus not recorded (False absence or false presence), e.g. in Colton & Swearer (2010);
- Objects obscured by other objects are not identified from the images, e.g. species growing under, or behind, other species (False absence);
- Species identification confuses one species with another (False presence);
- Differences between interpreters may occur;
- Methodological changes make it difficult to compare data between years.

Technological advances in underwater camera systems can be useful when developing criteria for the determination of GES. These include structural analyses of seafloor communities or assessing advances in fishing technology towards gears with less physical impact on the seabed and its associated biota (McShane *et al*., 2007; Carbines & Cole, 2009).

5.2.4 Image analysis

A large amount of data can be recovered from an image, ranging from anecdotal (description of a process); semi-quantitative data (e.g. degree of coverage of a fixed organism on a rock: a, none; b, low; c, medium; d, high; e, total); or more quantitative estimates. Quantitative data can be abstracted manually from a calibrated image by direct measurement and transformation. Computer software programmes are readily available to undertake this automatically. Computer image analysis systems can be extensive, covering the entire operation from image collection to output of analysis. These systems tend to be expensive and it is more common for the scientist involved in image analysis to have his own image input and storage system and to utilise readily available software ranging from professional analysis software to simpler image processing software (e.g. Adobe PhotoShop) and shareware (e.g. NIH Image). Most software allows on-screen measurement or, in more detailed systems, the filtering of images, automatic abstraction of shapes or parts and their automatic measurement.

Consideration should be given to the intended application of the final images, whether they are for internal use, scientific publication or public domain publication. For high quality output, high quality original images are needed as well as access to good printers. A grabbed video image may be acceptable on-screen, but it will not print in sufficient detail for publication. Storage and archiving are also important as different storage formats have different shelf-lives. Thermal print-outs from side scan sonar records will be affected by temperature, digital media by external voltage changes or magnetic fields. CDs quoted as having an indefinite shelf life may have a more limited shelf life. Important images should be kept at least in duplicate in separate locations in fireproof conditions. The quality of the material should be periodically checked and recopied if necessary onto newer standardised archiving formats. A comprehensive review of principles on this subject can be found in Glasbey & Horgan (1995).

Furthermore, recent advances in image processing and pattern recognition of plankton have made it possible to automatically or semi-automatically identify and quantify the composition of plankton assemblages at a relatively coarse taxonomic level (Benfield *et al.*, 2007). The Underwater Vision Profiler (UVP) can record abundances and size distributions of particles >100 μm and mesozooplankton in the water column up to 3000 m depth (Picheral *et al.*, 2010). The images are treated and analyzed in real time, and when the UVP is interfaced with a Conductivity Temperature Depth (CTD) sensor, physico-chemical information will complement the biological data. Considering these logistic advantages, the UVP is an ideal instrument for investigating the 'twilight' and deep-ocean zones, from meso- to global scales.

Image analysis can also be used to analyze zooplankton that has been sampled using traditional methods, e.g. by nets, pumps or water bottles. ZOOSCAN, for example, is an automatic image technique for zooplankton identification and enumeration (Gorsky *et al.*, 2010). However, the speed of analysis is comprised by the accuracy of identification and therefore, human visual recognition will still be needed to identify organisms at the species level. One of the advantages of this type of data analysis, however, is that a homogenous and secure digital zooplankton image data bank can be established and can be used for comparison of images across systems world-wide.

5.2.5 DIDSON

The DIDSON (Dual frequency IDentfication SONar) is a portable, high-resolution sonar system that uses acoustics (sound) to generate acoustic images with much more detail than conventional sonars. The images are comparable to those of echograms in hospitals and allow researchers to study fish behaviour, measure fish lengths and even determine species composition in turbid waters and even during the night. Currently, the DIDSON has been used in studies of fish behaviour in offshore wind farms and in studies of fish behaviour around obstacles (including the migratory routes of eel around sluices, Figure 5.3) to determine whether fish make use of escape routes.

The DIDSON has a range of 1–30 m and can function to a depth of 100 m (www.imares.nl). The use of the DIDSON for biodiversity monitoring, other than observing local behaviour, is not obvious at this stage.

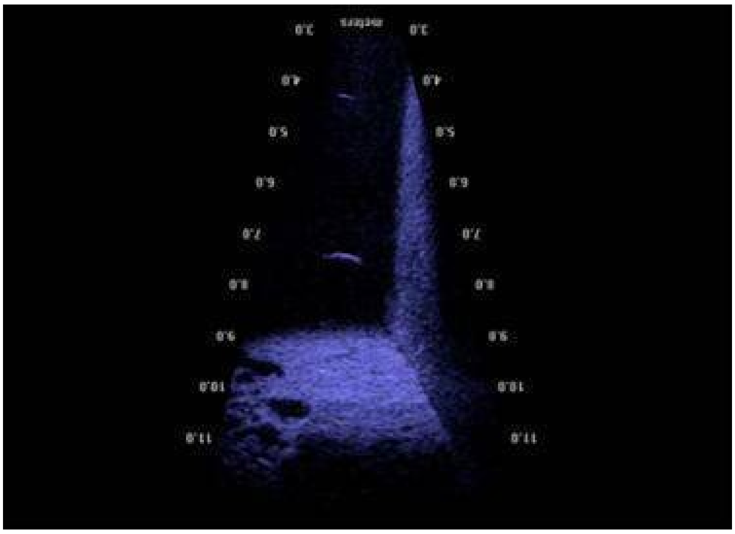

Figure 5.3. Video-still of DIDSON recording of an eel swimming towards a sluice (photo: Olvin van Keeken/ IMARES).

5.2.6 Electronic tagging

There have been several studies using electronic tags to better understand the behaviour, movements and migrations, and stock structure or a variety of marine taxa, including shellfish, elasmobranchs, teleosts and other marine vertebrates (Figure 5.4). The various technologies can include data storage tags (which may pop off), satellite tags and pinger tags (recorded by listening stations). A comprehensive review of these technologies and techniques is beyond the scope of the present report, and the reader is referred to, for example, Arnold & Dewar (2001).

In terms of biodiversity monitoring, such techniques can help better understand the behaviours and home range of individuals/stocks, including species of conservation and management interest (e.g. Hunter *et al.*, 2005; Stokesbury *et al.*, 2005; Metcalfe, 2006; Righton *et al.*, 2007; Wearmouth & Sims, 2009), and so then inform on potential merits of spatial management. It might also inform on the connectivity between different regions or sub-regions which are being monitored separately.

Figure 5.4. Harbour seal equipped with electronic tag (photo: Oscar Bos/IMARES).

5.2.7 Remote sensing

Remote sensing of the marine environment can provide high frequency and large-scale information on physical and biological processes such as oceanic fronts, eddies, primary production and toxic algal blooms. Eddies and fronts play an important role in supplying nutrients to the euphotic zone and, therefore, are often associated with enhanced biological production (Oschlies & Garçon, 1998). However, due to the high spatial and temporal dynamics of oceanic fronts, locating these features is not always straightforward. Novel analytical methods show improved precision in the visibility of these frontal features by combining frontal information from a sequence of satellite data (Miller, 2009). In addition to Advanced Very High Resolution Radiometer (AVHRR) SST fronts, the method has been validated on SeaWiFS chlorophyll fronts and is equally applicable to other ocean colour products for visualizing biological processes, such as algal blooms. As for all automated detection methods, the accuracy of detection must be validated by comparison with the true locations of fronts through the analysis of *in situ* transect sampling (Miller, 2009).

Remote sensing data can also be used for monitoring the spatial and temporal variations of the distribution of dominant phytoplankton groups. Recently, an algorithm has been developed to detect the major dominant phytoplankton groups from anomalies of the marine signal measured by ocean color satellites. This method, called PHYSAT, allows identifying nanoeucaryotes, *Prochlorococcus*, *Synechococcus* and diatoms (Alvain *et al.*, 2008). A synoptic approach combining the remote sensing data with novel *in situ* analysis methods (as carried out within the Interreg IVA DYMAPHY EU project) will allow functional groups of phytoplankton in the upper water column to be followed at a high spatial and/or temporal resolution. Phytoplankton functional groups can also be identified *in situ* by employing HPLC (Claustre *et al.*, 2004), or in some cases multispectral fluorometry in case of monospecific blooms by means of specific signatures (Beutler *et al.*, 2002) or flow-cytometric identification (Dubelaar & Geerders, 2004; Thyssen *et al.*, 2008).

5.2.8 Fishery acoustics

Fishery acoustics allows not only for the remote detection of schools and layers of (often commercial) pelagic fish such as herring or anchovy, but also provides information on other ecosystem components in the water column, such as zooplankton layers and larger-bodied non-schooling fish species.

The fundamental results of fisheries acoustics are well-constrained solutions to the so-called 'inverse problem' for marine organisms (Holliday, 1977a,b; Simmonds & MacLennan, 2005), where the number, size, and type of acoustic targets (fish and plankton) are estimated from acoustic volume backscatter measurements. The corresponding 'forward' problem involves computing the expected backscatter, given known numbers, sizes, and types of targets. A model that predicts the acoustic target strength is required in either case. Although inverse methods can in principle be used to estimate abundance, size, and composition of fish and zooplankton from backscatter measurements at multiple, appropriately selected frequencies (Holliday, 1977a,b), the problem is generally underdetermined (i.e. the number of unmeasured, or unknown, variables is greater than the number of measured, or known, variables) and complementary data collection is required to establish the size and species composition of dominant acoustic targets as well as to parameterize the scattering models. Complementary biological data are often collected with trawls or underwater cameras. Active underwater acoustic methods provide a means of collecting a wealth of ecosystem information with high space–time resolution (Table 5.1). Worldwide, fisheries institutes and agencies carry out regular acoustic surveys covering many marine shelf ecosystems, but these data are often underutilized.

In addition, increasing amounts of acoustic data collected by vessels of opportunity are becoming available. In a recent paper, Trenkel *et al*. (2011) reviewed and proposed indicators for assessing and monitoring zooplankton, population dynamics of fish and other nekton, and changes in diversity and food-web functioning (Table 5.2). The evaluation of new indicators and developing suitable reference points in different ecosystems are the current challenges.

Table 5.1. Overview of types of quantities and processes for which information can be extracted from active acoustic data, and their actual or potential use for ecosystem-based management (EBM) including the MSFD. MY: multi-year monitoring time series; E: experimental process study of limited duration. The space and time scales indicated are those relevant for EBM (Source: Trenkel *et al*. 2011).

Quantity or process of interest	Spatial scale	Time scale	Ecosystem components	EBM usage
Stock biomass or abundance index	Stock	MY	Fish, krill	Fish stock assessment; interpretation of single-species results; ecosystem models
Abundance index	Ecosystem	MY or E	Zooplankton, jellyfish, fish, species groups	Resource (prey) assessment; ecosystem models
Relationship between stock biomass and spatial spread	Stock	MY	Fish	Monitoring catchability
Predator-prey spatial relationships	Local or ecosystem	E	Fish-zooplankton interactions, marine mammals-zooplankton interactions	Identifying and monitoring food web structure; ecosystem models
Spatial distribution-physical habitat relationship	Ecosystem	E	Fish, zooplankton, hydrography/substrate	Ecosystem models; habitat mapping; climate change scenarios; spatial management

Table 5.2. Indicators and metrics for assessing environmental status of exploited marine ecosystems, derivable from acoustic data, and expected direction of change due to fishing and environmental changes (Source: Trenkel *et al*. 2011).

Category	Indicator/ Metric	Description	Theoretical basis	Effect of changes in environment	Effect of changes in fishing	Reference points
Species	*B*/*N*	Biomass/ abundance index	Yes	Unknown	Decrease	Biomass or abundance relative to historical situation
	L_{bar}	Mean length	Yes	Decrease by favourable recruitment	Decrease	Unknown
	SA	Spreading area	Yes	Depends on species	Decrease	Relative to historical situation
	$B_{zooplankon}$	Timing of zooplankton peak biomass	Yes	Spatio-temporal shifts in peak abundance	None	Relative to prey abundance timing
Diversity	acoustic diversity	Diversity of acoustic species	Empirical	Depends on definition of acoustic species	Depends on definition of acoustic species	Relative to historical situation
	acoustic spectrum	Slope of acoustic energy spectrum	Empirical	Depends on definition of acoustic groups	Depends on definition of acoustic groups	Relative to historical situation
	acoustic dominance	Acoustic energy by frequency	Empirical	Unknown	Unknown	Relative to historical situation
Food web	B_k	Biomass of key group	Empirical	Depends on trophic position	Depends on trophic position	Relative to historical situation
	GIC	Predator-prey global index of co-location	No	Unknown	Decrease	Relative to historical situation

5.2.9 VMS and fisher behaviour

The Vessel Monitoring through Satellite (VMS) system sends information on the position of fishing vessels to authorized data managers. By analysing the data it is possible to reconstruct when and where a ship was fishing, show the importance of specific areas for the fishery, get insight in fishers fishing behaviour and estimate pressures on e.g. the bottom-living animals in that area. One problem in relation to VMS data is the very limited and restricted availability to the research community of aggregated and standardised international effort data by metier.

5.2.10 Deck cameras /CCTV

Technology-based fishery monitoring, or electronic monitoring (EM), has emerged as an alternative to human observers on board fishing vessels and is being applied in a variety of fisheries. Deck cameras and CCTV systems are a central part of this approach. They have recently been used for implementing management measures such as reporting of total catches in Danish fisheries (Kindt-Larsen *et al*., 2011) or verifying fishers self-reporting of catches in a hook-and-line fishery in British Columbia (Stanley *et al*., 2011). Comparison between CCTV-based catch numbers by species and those recorded by onboard observers generally shows good agreement, but also some differences (Ames *et al*., 2007).

Although the current motivation behind the development of CCTV systems is primarily enforcement and collection of catch data for quota management, information on conspicuous species, including unmanaged, protected species, or species of conservation interest (e.g. marine mammals, elasmobranchs), could be collected which could be useful for biodiversity monitoring and management. Using these data would require development of appropriate data treatment methods given the strong behavioural component (where fishing takes place, what gear is used etc.) and absence of any statistical sampling design in this "data collection" procedure.

5.2.11 Genetic tools and barcoding

Genetic techniques to identify specimens to the species level are developing rapidly and are very promising, but are not used yet for regular biodiversity monitoring purposes.

DNA barcoding is a taxonomical method that uses a short unique sequence (genetic marker) in an organisms DNA to identify species. DNA barcoding cannot be used to determine variation within species. Databases such as the Marine Barcode of Life already contain over 6100 barcoded marine species worldwide, including 50% of known elasmobranch species and 35% of known fish species. In contrast, less than 1% of benthic invertebrates have been identified. The rapid development of these techniques allows researchers to analyze as many as 15 000 different sequences in a few hours' time.

DNA barcoding can be used for the analysis of the species composition of zooplankton, benthic meiofauna samples, young stages of benthos on settlement plates, samples of fish eggs and larvae, and stomach contents, although the quality of such analyses can be dependent on preservation techniques etc. The technique can also be used to check for non-indigenous species in ballast water or growing on ship hulls, provided that the barcodes are known (www.marinebarcoding.org; Van Pelt, IMARES, Pers. Com.).

5.2.12 Internet-based identification and biodiversity data portals

Internet based data portals providing information on species identification, taxonomy, abundance and distribution have become increasingly important for large-scale biogeography and biodiversity studies.

An important example of an open access data portal providing identification keys includes the **Marine Species** Identification Portal (www.species-identification.org). To date, this portal compiled information on 9900 marine species and 5553 higher taxa, most of which with a description and one or more illustrations and total of 7941 taxa are keyed out in 52 identification keys. The information was assembled over a period of 10 years by a global network of collaborating taxonomists that was started with UNESCO support.

Another example of a comprehensive database, in this case on species taxonomy, is the World Register of Marine Species (WoRMS; www.marinespecies.org). In contrast to most data portals, the content of WoRMS is controlled by taxonomic experts, not by database managers; however, quality control is still carried out by the editorial management system (Appeltans *et al.*, 2012).

Large-scale biogeographic studies often require assembling extensive biological datasets from disparate sources. An increasing number of institutes are now sharing their data through user-friendly and open access data portals where information on marine species can be freely downloaded. Some of these online data portals also provide interactive tools to visualize species occurrences for a chosen spatial extent and even predict species distributions using ecological niche based models.

The Ocean Biogeographic Information System (OBIS; www.iobis.org), with EurOBIS as the European node (www.marbef.org/data/eurobis.php) for example, is a web-based provider of global geo-referenced information on marine species, with online tools for visualizing relationships among species and their environment. This database was created by the Census of Marine Life and is now part of the Intergovernmental Oceanographic Commission (IOC) of UNESCO, under its International Oceanographic Data and Information Exchange (IODE; www.iode.org) programme. A more comprehensive open access data portal is the Global Biodiversity Information Facility (GBIF; www.gbif.org) including information on marine and terrestrial biodiversity. Through a global network of countries and organizations, GBIF promotes and facilitates the mobilization, access, discovery and use of information about the occurrence of organisms over time and across the planet. More specialist data portals focus on single groups, such as fish (Fishbase; www.fishbase.org), but can provide extensive information and data on taxonomy, geographical distribution, biometrics and morphology, behaviour and habitats, ecology and population dynamics as well as reproductive, metabolic and genetic data on a large spatial scale. Moreover, interactive tools can used for calculating trophic pyramids, fishery statistics and perform biogeographical modelling.

AquaMaps (www.aquampas.org) is a species distribution model available as an online web service that generates standardized range maps and the relative probability of occurrence within that range for currently more than 9000 marine species from available point occurrences and other types of habitat usage information (Kaschner *et al.*, 2008). By overlaying AquaMaps predictions for a subset of individual species (namely 115 marine mammals), a global map of biodiversity patterns was produced that shows the co-occurrence of predicted hotspots of marine mammal species richness and off-shore seamounts.

Although internet-based sources can provide easily accessed and comprehensive data, it must be remembered that not all internet-based information and data sources are thoroughly peer-reviewed, and so outputs may need to be critically evaluated.

5.3 Improved opportunities of data collection

5.3.1 Better use of existing scientific surveys

More work of relevance to biodiversity monitoring could be undertaken on existing surveys, including those internationally-coordinated by ICES (e.g. through IBTSWG and WGBEAM). Such work could include collecting tissue samples for genetic studies, data on other ecosystem components (sea birds, marine mammals, benthos), and other aspects of ecosystem structure and function (acoustics, oceanography, feeding habits). Such issues have been discussed in several other ICES reports (e.g. the Working Group on Integrating Surveys for the Ecosystem Approach) and are not detailed here.

5.3.2 Improved use of platforms of opportunity

Platforms of opportunity (e.g. ferries and merchant ships) can provide a cost-effective means to collect large spatial oceanographic data by on-board observers or by carrying scientific instruments (Evans & Hammond, 2004; Kiszka *et al.*, 2007). To date, the biological data collected in Europe using platforms of opportunity include phytoplankton, zooplankton, seabirds and marine mammals. For example, the European

FerryBox Network incorporates 11 research institutes from eight countries and deploys automated sensors for measuring biological, chemical, and physical variables, which are attached to commercial ferries. Other projects include the Continuous Plankton Recorder (CPR) survey, the largest plankton monitoring programme in the world. The plankton sampling instrument (Figure 5.5) is towed from merchant ships on their normal sailings and has monitored the presence or abundance of more than 400 plankton species on a monthly basis over the North Atlantic since 1946. After a CPR has been towed, it is returned to the laboratory at SAHFOS, UK, for analysis to obtain estimates of chlorophyll *a* concentration using a "greenness index" known as the Phytoplankton Colour Index (PCI) and zooplankton abundance and species composition. The CPR's long time-series and extensive spatial coverage and unchanged methodology has enabled the development of statistically significant complex multivariate indicators encompassing many levels of ecosystem state, structure, and functioning (McQuattors-Gollop *et al.*, 2010).

Figure 5.5. Continuous Plankton Recorder on a Dutch ferry (photo Robbert Jak/ IMARES).

5.3.3 Better use of fishery-dependent data sources

The EU-funded onboard observation schemes, whose primary purpose is to collect information on discards, might provide a valuable data source for biodiversity monitoring. For example, spatial indices from the French onboard observation program in the Bay of Biscay were compared to those derived from survey data (see Section 2). The two sources showed a good overall agreement. Further studies are needed to evaluate the potential and limits of onboard observations and other fishery-dependent data for biodiversity monitoring.

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Annex 2: WGBIODIV draft resolution for the next meeting

The **Working Group on Biodiversity Science** (WGBIODIV), chaired by Simon Greenstreet*, UK, will meet at ICES headquarters, Copenhagen, Denmark, February 2012 (TBA) to:

a) Further review the development of indicators of biodiversity, including those supporting the Marine Strategy Framework Directive (MSFD) and other initiatives (e.g. Convention on Biological Diversity), including:

   i. undertaking case studies of those 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. investigate the utility of other potential biodiversity metrics that may inform on other aspects of biodiversity and ecosystem structure and function,

   iii. consider how metrics and indicators for various facets of marine biodiversity might be integrated to derive more regional and holistic assessments of environmental and biodiversity status,

   iv. investigate the spatial and temporal variation of biodiversity indicators, and how that may inform on monitoring programmes and their design, indicator reporting and management.

b) Examine spatial/temporal aspects of the biodiversity of areas of particular interest to better understand and define biodiversity hotspots.

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 (MSFD).
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. ToR (a) supports current work being undertaken by various ICES nations in support of the MSFD, and ToR (b) supports the application of biological information into spatial management.
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