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Executive summary

Biodiversity has become an increasingly important element of ICES' work and is one of the research topics identified in the ICES Science Plan as being of strategic importance to the advisory needs of ICES. The European Commission's (EC) recent Marine Strategy Framework Directive (MSFD) also 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. A range of ICES Expert Groups are currently involved in particular aspects of marine biodiversity, and WGBIODIV aims to provide the ICES community with an improved capacity to coordinate integration and synthesis of biodiversity information.

There are gaps in current research expertise and data management that affect the ability of ICES to provide effective management advice in terms of biodiversity. One of the important issues to be addressed is what components of marine biodiversity are currently being surveyed by the ICES and wider scientific communities, and at what spatial and temporal scale. WGBIODIV has provided an overview of the current field programmes assessing some of the major marine taxa across the ICES ecoregions, and highlighted some of the relevant advantages, limitations and caveats in terms of how such data can be applied to biodiversity science (see Section 2). Whereas there is a long history of coordination across the ICES community for surveying various marine fishes (e.g. through trawl surveys), the spatial and/or temporal extent for surveys examining other marine taxa is often more limited.

There is a growing interest in the use of indicators to inform on the state of the marine environment, including biodiversity. Data from existing, broad scale surveys of the wider continental shelf will likely be one of the main resources used by the ICES community in the calculation of metrics and indices of biodiversity. WGBIODIV, however, considered that there were various elements of macroecology that need to be better considered prior to using such data for indicators (see Section 3). These issues include the processes regulating the number of species; species and stock concepts; succession, community regulation and 'baseline' conditions; habitat complexity; resilience; rarity; extinction and extirpation; and large scale patterns of diversity in the sea.

There are a wide variety of indices and metrics that may be considered for the development of 'biodiversity indicators', including species-specific metrics; traditional multi-species community/assembly metrics; taxonomic diversity; functional diversity; size-based and food-web or trophic indicators. An overview of such metrics is given in Section 4.

1 Introduction

1.1 Background

The Study Group on Biodiversity Science (SGBIODIV) first met in 2007 in Belgium (ICES, 2007a), and reported on possible contributions by ICES on biodiversity science, especially in terms of how such knowledge on biodiversity science could be used in the Ecosystem Approach to Management (EAM).

SGBIODIV met again in Belgium the following year (ICES, 2008) in order to define 'biodiversity science', to report on the remit of the group, to review current and emerging marine biodiversity initiatives, and to provide an overview of how other ICES Expert Groups contributed to biodiversity science.

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

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 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". The remit of the group is "to recommend mechanisms that will advance ICES' capacity to understand and provide advice on the effects of human activities and natural change on marine biodiversity".

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

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

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

graphic and climatic conditions”, although several of the other descriptors are also biodiversity-related.

The EC has also recognised the importance that “monitoring methods are consistent across the marine region or subregion so as to facilitate comparability of monitoring results” (CEC, 2008). Consequently, ICES will likely be involved in ensuring standardised sampling (e.g. through the survey groups within the SCICOM Steering Group on Ecosystem Surveys Science and Technology (SSGESST)) and analyses of such data that may be undertaken by various ecology and other Expert Groups.

1.2 Terms of Reference

The Study Group on Biodiversity (SGBIODIV), chaired by Jim Ellis, UK, will be re-named Working Group on Biodiversity (WGBIODIV) and will meet at IPIMAR, Lisbon, Portugal, 22–26 February 2010 to:

- a) Contribute to the Strategic Initiative on Biodiversity led by SSGSUE¹. In doing so, develop a working plan on biodiversity in ICES ecoregions for dominant marine taxa and identify relevant data sets for examining temporal and spatial change.;
- b) Review existing approaches to the development of biodiversity indicators;
- c) Synthesise biodiversity information from other EGs for specific advice requests;
- d) Report by March 15 on potential contributions to the high priority topics of ICES Science Plan by completing the document named "SSGEF_workplan.doc" on the SharePoint site. Consider your (WGBIODIV) current expertise and rank the contributions by High, Low or Medium importance;
- e) Prepare contributions for the 2010 SSGEF session during the ASC on the topic areas of the Science Plan which cover: Individual, population and community level growth, feeding and reproduction; The quality of habitats and the threats to them; Indicators of ecosystem health.

1.3 Participants

The following participants attended the meeting or contributed by correspondence (denoted *). Contact details are included in Annex 1.

Maria Fatima Borges	Portugal
Anik Brind'Amour	France
Fátima Cardador	Portugal
Corina Chaves	Portugal
* Wenche Eikrem	Norway
Jim Ellis	UK (England & Wales)
Ivone Figueiredo	Portugal
Maria José Gaudêncio	Portugal
Miriam Guerra	Portugal
Reinier Hille Ris Lambers	Netherlands
Ingo Narberhaus	Germany

¹ Now a SCICOM initiative

Heye Rumohr	Germany
Melanie Sapp	UK (England & Wales)
* Michaela Schratzberger	UK (England & Wales)
Francisco Velasco	Spain
Ana Rita Vieira	Portugal
Maria Wlodarska-Kowalczyk	Poland

1.4 Summary of Working Documents and presentations

The following presentations were made during the meeting:

Sapp, M.: Microbial diversity and its importance for ecosystem functioning

Microorganisms are those organisms that are not visible to the naked eye or < 20 µm, and include viruses, archaea, bacteria and picoeukarya. Although microbes are important in ecosystem functioning (e.g. they are involved in biogeochemical cycling, are integral in food webs like the microbial loop etc.), less than 0.5% of the estimated 2–3 x 10⁹ microbial species have been identified.

Włodarska-Kowalczyk, M.: Surrogates in assessment of patterns of benthic distribution and diversity

Surrogates are taxonomic units that are more easily determined than, but which still correlate strongly to, species-level community patterns, so facilitating assessments (e.g. for pollution monitoring) that require less taxonomic expertise (i.e. are less time consuming and less expensive). Surrogate methods may involve only using data to a higher taxonomic level (e.g. genus or family), or using a particular faunal group (e.g. polychaetes).

Cardador, F. & Chaves, C.: Portuguese Groundfish surveys

Portuguese groundfish surveys started in 1979, with an average of two surveys per year, in winter, summer and autumn. The winter series stopped in 2008, the summer series in 2002 and the autumn series continues to date (1979–2009). The survey description was presented, concerning main objectives, area of coverage, sampling design, vessel and gear, onboard methodology and target species. During 2005–2009 autumn surveys a total of 169 species were identified, including 99 fish, 23 crustaceans, 15 cephalopods, 13 echinoderms, and seven bivalves, with the remaining species including a variety of ascidians, cnidarians, gastropods and polychaetes. The bottom trawl net used in this survey has rollers in the groundrope, and so the benthic species are not well sampled. Several Portuguese studies on biodiversity and fish assemblages have been made using data from these surveys (Gomes *et al.*, 2001; Morgado *et al.*, 2001; Sousa, 2006; Sousa *et al.*, 2005, 2006, 2009).

Working Documents provided were:

Silva, C. & Leotte, F. (2007). Portuguese *Nephrops* in FU 28+29 (CTVS) survey. 6pp.

This WD was originally presented at the 2007 International Bottom Trawl Survey Working Group Meeting, describing the Portuguese Crustacean surveys directed to *Nephrops* in Portuguese waters which have been conducted since 1981 to estimate the relative abundance of *Nephrops*, as well as other crustacean species (e.g. deepwater rose shrimp, red shrimp). During 2005–2009 a total of 183 species were identified, including 98 fish, 45 crustaceans,

16 cephalopods, 9 echinoderms and 8 gastropods, with the remaining species including a variety of bivalves and cnidarians.

This WD can be found in Annex 9 of ICES (2007b)

1.5 Theme session at the 2010 ASC

It was suggested during the 2009 meeting of SGBIODIV that a Theme Session on Marine Biodiversity should be convened at the 2010 ICES ASC, with 2010 being especially timely given the 2010 time frame indicated by the CBD. The United Nations has also declared 2010 to be the International Year of Biodiversity.

This proposal was initially drafted by SGBIODIV members intersessionally and was accepted. Theme Session Q “Marine Biodiversity – the science and management needed to meet 2010 commitments” will be convened by Jake Rice (Canada), Heye Rumohr (Germany), Carlo Heip (Netherlands), Paul Snelgrove (Canada), and Thomas Noji (USA).

The Theme session was discussed during the course of the meeting, and it is planned that a joint paper will be submitted, in addition to any individual papers.

1.6 References

- CEC. 2008. Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive).
- ICES. 2007a. Report of the Study Group on Biodiversity Science (SGBIODIV), 9–11 May. MHC:11; 31 pp.
- ICES. 2007b. Report of the International Bottom Trawl Survey Working Group (IBTSWG), 27–30 March 2007, Sète, France. ICES CM 2007/RMC:05. 195 pp.
- ICES. 2008. Report of the Study Group on Biodiversity Science (SGBIODIV), 11–14 March 2008, Gent, Belgium. ICES CM 2008/MHC:06; 71 pp.
- ICES. 2009. Report of the Study Group on Biodiversity Science (SGBIODIV), 17–20 March 2009, Wilhelmshaven, Germany. ICES CM 2009/MHC:05. 51 pp.
- Gomes, M. C.; Serrão, E. and Borges, M. F., 2001. Spatial patterns of groundfish assemblages on the continental shelf of Portugal. *ICES Journal of Marine Science*, 58: 633–647.
- Morgado, C.; Chaves, C.; Murta, A.; Cardador, F. and Azevedo, M. 2001. Biodiversity of the Portuguese continental waters based on groundfish survey data, Poster presented to EURESCO - Conference 2001 - Biodiversity of Coastal Marine Ecosystems -Corint, Greece, 5–10 May 2001.
- Sousa, P., 2006. The demersal community off Portugal: a study on zoogeography, abundance, and biodiversity. *Dissertação de Doutoramento em Biologia (Especialidade de Biologia Populacional)*. Faculdade de Ciências da Universidade de Lisboa. 196 p.
- Sousa, P., Azevedo, M. & Gomes, M.C. 2005. Demersal Assemblages off Portugal: mapping, seasonal, and temporal patterns. *Fisheries Research*, 75: 120–137.
- Sousa, P., Azevedo, M. & Gomes, M.C. 2006. Species-richness patterns in space, depth, and time (1989–1999) of the Portuguese fauna available to bottom trawl. *Aquatic Living Resources*, 19 (2): 93–103.
- Sousa, P.; Cardador, F. 2009. *Manual de Análise de Dados - Zoogeografia e biodiversidade*. Ed. L-IPIMAR, 59 p. ISBN: 978-972-9372-35-3.

2 Biodiversity in the ICES area

2.1 Introduction

This section of the report summarises some of the current knowledge of the 'biodiversity' of the dominant marine taxa in the ICES area and eco-regions (Figure 2.1) and addresses ToR (a).

2.2 Microbial diversity

2.2.1 Introduction

Biodiversity encompasses all biota ranging from viruses and unicellular to multicellular organisms. Here, we introduce microbial diversity as an essential component of biodiversity. Generally, the denotation "microorganism" describes objects not visible to the naked eye or being $<20\ \mu\text{m}$ in size. This encompasses viruses, prokaryotes and small eukaryotes, such as flagellates. Their diversity can be described based on 16S and 18S rRNA gene sequences resulting in a phylogenetic tree of life presenting high microbial diversity (Woese *et al.* 1990).

Overall, microorganisms may comprise ca. 60% of the earth's biomass, with an estimated diversity of $2\text{--}3 \times 10^9$ microbial species. Microbial communities differ from those formed by larger, multicellular organisms mainly with respect to the characteristic scales of time and space. These differences are primarily determined by the rates of growth, large population sizes resulting in a high capacity of genetic diversity (Whitman *et al.* 1998), the ratio of surface to volume of the cells and the interaction with physical properties of the environment. Furthermore, high redundancy within microbial communities can be observed (Langenheder *et al.* 2006), which might play an important role for ecosystem resilience.

How the microbial role in ecosystems was perceived has changed dramatically in recent decades. In the 1960s, bacteria in marine ecosystems were exclusively ascribed to decomposition processes (Strickland 1965). A decade later, Pomeroy (1974) considered photoautotrophic nanoplankton to be the major primary producer in marine food webs and stated that particulate and dissolved organic matter (POM and DOM) would be important food sources which would be consumed primarily by heterotrophic bacteria. This idea was taken up by the concept of the microbial loop presented by Azam *et al.* (1983) in which energy within DOM and POM assumed to be lost from the trophic system is channelled back to the pelagic food chain. Recently, research in marine microbial ecology has further increased our knowledge regarding the role of microbes (e.g. Finlay *et al.* 1997; Arrigo 2005; DeLong & Karl 2005; DeLong *et al.* 2006; Revsbech *et al.* 2006; Wallenstein *et al.* 2006; Kirchman *et al.* 2007), including for the biogeochemical cycling and channelling of energy (Duffy & Stachowicz 2006), thus providing irreplaceable ecosystem services. It is assumed that ecosystem functioning is strongly coupled with microbial activity (Finlay *et al.* 1997) and it is likely that microbial diversity is related to changes of ecosystem state.

Recent developments in methodology resulted in promising prospects delivered by high resolution fingerprinting methods that enable researchers to predict temporal patterns in community structure (Fuhrmann *et al.* 2006). It is therefore suggested that inclusion of microbial community-based studies in marine monitoring will contribute powerful information for an evaluation of ecological limits with regard the sustainability of principle ecosystem processes.

2.2.2 Current ICES expertise

Until recently, an ICES Study Group on Phytoplankton and Protist Taxonomy existed, but this was dissolved in 2008, as was the Working Group on Phytoplankton Ecology (WGPE). However, the Working Group on Phytoplankton and Microbial Ecology (WGPME) was established, and will be meeting in March 2010. This will develop an action plan focussing on phytoplankton and other unicellular microbes within the ICES Science Plan and form links to other Working Groups, although WGPME may address pelagic ecosystems more than benthic ecosystems.

2.2.3 Availability of survey data

Currently, there is a lack of consistent monitoring programmes for microbial community studies. Further input regarding data acquisition was given by the Working Group on Phytoplankton and Microbial Ecology summarising current long time series for phytoplankton mainly (<http://wgpme.net>).

2.2.4 Data quality issues

The study of marine microbes started exclusively with cultivation approaches which were supplemented later by direct counting (Jannasch & Jones 1959; ZoBell 1946). It became obvious that the data obtained displayed great variability known as “the great plate anomaly” (Staley & Konopka 1985). It is generally accepted that only few marine microorganisms can be cultivated. Therefore, phylogenetic diversity of viruses and prokaryotes is studied mainly with molecular tools. On this basis, diversity-related studies have been utilizing methods like cloning and sequencing of specific genes as well as community fingerprinting (e.g. Pulsed Field Gel Electrophoresis PFGE, Amplified Ribosomal DNA Restriction Analysis ARDRA, Ribosomal Intergenic Spacer Analysis RISA, Single Strand Conformation Polymorphism SSCP, Terminal Restriction Fragment Length Polymorphism T-RFLP and Denaturing Gradient Gel Electrophoresis DGGE), providing information on the diversity and spatio-temporal dynamics of microbial communities.

In addition to these semi-quantitative methods, fluorescent *in situ* hybridization provides quantitative information on microbial populations. For eukaryotic microorganisms microscopic techniques including scanning electron microscopy are widely used for identification. Recently, molecular methods, often based on the 18S rRNA gene, were applied to gain comprehensive information on these communities.

Considering the importance of functional diversity, the study of genetic heterogeneity of functional genes (Priemé *et al.* 2002) is adding value to ecosystem state assessments. At present, these methods are, however, restricted to available sequences of relevant genes. The major processes covered are nitrogen fixation, nitrification and denitrification.

Additionally, recent technical developments such as shotgun sequencing (Venter *et al.* 2004) improved the characterization of microbial communities considerably resulting in a better reflection of the community and the identification of rare species. Metagenomic studies were shown to provide information on a functional level without restriction by the fingerprinting method applied (Schloss & Handelsman 2005). However, data acquisition is time-consuming and labour-intensive with quality depending on the experience of researchers. Additionally, it should be taken into account that diversity patterns obtained with different methodological approaches might not be directly comparable. Currently, there are no quality control schemes based on certified reference materials implemented for the study of microorganisms.

2.2.5 Taxonomic issues

Although microorganisms can be found ubiquitously, their detection and identification is largely constrained by methodology. Furthermore, the species concept for microbes is still under debate, as are speciation processes and current measures used to describe new species (Achtmann & Wagner 2008). Currently, new microbial species are defined based on a variety of characteristics including phenotype and genotype. One characteristic is based on reciprocal, pairwise DNA re-association values where new species should display < 70% similarity in DNA-DNA hybridization experiments. Also sequence similarity of $\leq 98.7\%$ of the 16S rRNA gene indicates different species and correlates with < 70% similarity in DNA-DNA hybridisation (Stackebrandt & Ebers 2006).

2.2.6 Capacity for ICES to include microbes within 'biodiversity' advice

Existing monitoring programmes could be used to collect samples for microbial community analyses. However, further laboratory based analyses would need to be funded to increase the knowledge base in this field. It is therefore important to strengthen the link with the Working Group on Phytoplankton and Microbial Ecology to obtain more information on existing long term data sets within the ICES region.

2.3 Meiofauna

2.3.1 Introduction

Biodiversity in the marine environment is typically assessed by reference to the status of and changes in the structural, and increasingly functional, attributes of biological communities. The success of any biodiversity research or monitoring programme in detecting the extent and degree of change is thus determined by a number of factors, including the sampling strategy, the target faunal groups, the biological parameters recorded and the methods employed to analyse and interpret the collected data.

For reasons of convenience, most biodiversity studies have traditionally targeted large visible organisms that can readily be counted and identified. Consequently, there have been few multidisciplinary studies including meiofauna, an assemblage of marine benthic invertebrates with dimensions between 500–1000 and 32–63 μm , in national and international programmes. This size spectrum separates a discrete group of organisms whose morphology, physiology and life history characteristics have evolved to exploit the interstitial matrix of marine sediments. The meiofauna, representing 23 phyla, are phylogenetically more diverse than other components of the marine biota. As a discrete benthic component and, as a result of their high abundance and diversity, ubiquitous distribution, rapid generation times and fast metabolic rates the meiofauna play an important role in biodiversity and ecosystem function and thus the state of meiofauna assemblages may reflect overall benthic health (Giere 2009, Table 2.1).

2.3.2 Current ICES expertise

In 2006, SGBIODIV summarised the contribution of various ICES Expert Groups to research activities or information on the components of marine biodiversity and to the management of the components of marine biodiversity (ICES, 2009a). The majority of relevant groups reviewed scientific knowledge on structural and functional aspects of species focussing on alien and native, target and non-target species and their populations. Community structure of major ecosystem components has been re-

viewed as have inter-specific competition and function and productivity of selected taxa, but none of these reviews explicitly dealt with meiofauna. This is a combined reflection of (a) the absence of specific requests for advice on meiofauna-related issues, (b) in the case of more general requests on biodiversity- and ecosystem-related issues the limited information available on meiofauna diversity and, consequently, (c) the lack of meiofauna experts within the ICES community.

2.3.3 Availability of survey data

In 2005, EU Network of Excellence on Marine Biodiversity and Ecosystem Functioning (MarBEF) funded the development of an integrated database including information on the structure, diversity, dynamics and functional role of marine meiofauna (MANUELA).

This database (<http://www.marbef.org/projects/Manuela/description.php>) currently comprises 83 datasets (approximately 1300 stations and 140 000 distribution records, Table 2.2, Figure 2.2). The datasets range from the deep-sea to the coastal zone and from the Arctic to the Antarctic, with a focus on the North-East Atlantic region and the North Sea. The advantages of the integrated database include standardisation of species lists (see Section 2.3.4), data quality control (see Section 2.3.5) and bringing together large amounts of information varying over space and time (Vandepitte *et al.* 2009).

Traditionally, monitoring of meiofaunal diversity has included studies of species assemblages and the majority of these studies have been performed over a limited spatial and temporal scale. A number of surveys have been carried out in the ICES area, most of them in polluted fjords and coastal areas affected by human activities, but also in less influenced offshore areas.

One of the most spatially extensive data sets is available from the ICES North Sea Benthos Survey in 1986. Although some information describing meiofauna densities and harpacticoid copepod assemblage structure from these data has been published by Heip *et al.* (1992) and Huys *et al.* (1992), there has been no comprehensive diversity assessment of other meiofaunal taxa. For the North Sea, such information is only available from the coastal areas of Belgium, the Netherlands, Germany and the United Kingdom (Heip *et al.* 1990, Vanreusel, 1990, Vincx 1990, Steyaert *et al.* 1999, Schratzberger *et al.* 2000, Schratzberger *et al.* 2006, 2007, Vandepitte *et al.* 2009).

Long-term programmes with repeated studies have been established in few locations only. Kristensen (2005), for example, collected bottom samples on the Faroe Bank on five occasions between 1989 and 2001 for the qualitative analysis of meiofauna. Samples were collected between 90 and 1040 m and the sediments varied from basalt cobble and coarse shell-gravel to fine carbonate silt and mud.

2.3.4 Data quality issues

Various laboratories engaged in meiofaunal research follow internal quality control procedures to ensure that consistent data of a high standard are obtained. A set of Standard Operating Procedures (SOPs) generally covers sample collection, sample processing, laboratory procedures, equipment and supplies, quality control procedures, safety, data quality and reporting requirements. The application of such quality assurance protocols (where they exist) enables the results of separate investigations to be compared from year to year and evaluated holistically across a geographical area.

At present there are no nationally or internationally agreed quality control schemes for meiofauna through the analysis of suitable and certified reference materials. Consequently, the integration of meiofauna data from various sources requires post-identification standardisation as was carried out when the MANUELA database was created (see Section 2.3.3, Vandepitte *et al.* 2009). Standardisation of taxonomic names, for example, was achieved by matching all the taxonomic names against the European Register of Marine Species (ERMS). Further standardisation efforts focused on geographical names, sampling methodology and abiotic variables.

2.3.5 Taxonomic issues

Meiofauna are especially sensitive to environmental change, but their taxonomic diversity is often difficult to characterise because of a lack of available specialist taxonomic knowledge of the many different phyla involved (Royal Society 2003). Nematodes in particular are often neglected in meiofaunal surveys owing to the small size of each individual, the relative paucity of easy morphological characters and the difficulty in identifying juvenile and other stages. Progress has been made in providing the means for meiofaunal work by non-specialists with the publication of pictorial keys for the identification of meiofauna species in the Synopses of the British Fauna (Platt & Warwick 1983, Platt & Warwick 1988, Huys *et al.* 1996, Warwick *et al.* 1998). These keys mark a major advance in the facility to identify (British) marine nematodes and harpacticoid copepods. They have stimulated greater interest in meiofauna studies which in turn has resulted in an increasing number of meiofauna experts world-wide, specialised particularly in the identification of the hard-bodied taxa.

Modern DNA-based methods offer the prospect of simple and unambiguous surveys of diverse but taxonomically difficult groups, although few attempts have been made to apply these to marine meiofauna (Bhadury *et al.* 2006). This is partly a result of the labour-intensive nature of the isolation of meiofauna individuals and the generation of molecular sequences. Consequently, very few DNA sequences currently exist from properly identified meiofaunal specimens in public databases. These are essential to establish the relationship between historical knowledge based on morphology and Linnaean taxonomy, and the new molecular taxonomy.

Recently, easy-to-use species identification tools have been made widely available via the world-wide web such as the NeMys database developed by Ghent University (Deprez 2005, <http://nemys.ugent.be>). NeMys is a generic online species information system, storing information of various taxa including nematodes. The database stores data on morphology, biogeography, taxonomy, literature, pictures, collections and molecular aspects in one single dataset, thereby creating a large group of possible end users and allowing data analysis and comparison for a specific taxonomic group from a variety of approaches. NeMys data is shared with a number of global and regional biodiversity information portals such as the European Register of Marine Species (ERMS), the Ocean Biogeographic Information System (OBIS) and the Global Biodiversity Information facility (GBIF).

2.3.6 Capacity for ICES to include meiofauna within 'biodiversity' advice

The ICES Science Plan and subsequent restructuring of ICES science reflects the appreciation that management decisions aimed at protecting the marine environment against adverse effects of human activities and conserving biological diversity require a more holistic approach than hitherto. Schratzberger *et al.* (2000, 2004) assessed environmental quality around the UK coast through the examination of benthic assem-

blage structure at inshore and offshore locations. They concluded that the use of both macrofauna and meiofauna techniques in routine monitoring not only provided complementary information on environmental conditions and greater flexibility to meet site-specific study requirements but also, most importantly, widened the scope for evaluation of the status of the benthic ecosystem as a whole.

As indicated in preceding subsections, whilst the inclusion of meiofauna within holistic biodiversity-related advice is essential, at present this could be based on findings arising from individual studies performed over a limited spatial and temporal scale.

2.4 Benthic infauna

2.4.1 Introduction

The fauna of the sediments of the ICES area have been subject to many studies over the last 100 years or so, partly due to the early, traditional interest in marine zoology, subsequent studies of 'fish food' and then in examining the effects/impacts of anthropogenic disturbance on the sea floor (Rees *et al.*, 2008). The benthic infauna include a variety of taxa, the dominant groups being annelids and other worm-like taxa (platyhelminths, nemerteans, priapulids, echiurans, sipunculids), as well as various crustacean, mollusc and echinoderm taxa.

2.4.2 Current ICES expertise

The current expertise of ICES on benthic infauna and benthos in general was very much created and collected by the Benthic Ecology Working Group (BEWG). After this Expert Group was established in 1981, they aimed at a joint sampling programme of North Sea benthos which could incorporate the existing UK and Scottish benthos data collected by Eleftheriou and others. This joint sampling project was conducted in 1986 with great success (and no external funding) despite all the problems that are included in international comparative work (e.g. different national surveys, methodologies and taxonomic standards) that led to time consuming activities to construct a single species list with accepted names. These were activities from which the current databases of ICES, MacroBEN and ERMS still profit. This joint sampling approach was repeated in 2001 in an opportunistic way by using available data from national benthos sampling programmes. The results and data have been published in peer reviewed journals and as an ICES Cooperative Research Report (Rees *et al.*, 2007). The epifauna of the North Sea was also jointly collected in 2001/2002 and published in joint reports that went into recent databanks

ICES data centre concentrated for years to collate benthic data sets from European water. This was fostered by the task to maintain the HELCOM database covering the Baltic HELCOM Monitoring scheme, a piloting environmental surveillance scheme that acted as model for many other monitoring projects.

BEWG has also addressed the marine phytobenthos (i.e. algae and phanerogames) in recent years, to build up the needed expertise for coastal management questions and coastal ecological questions.

Additionally, BEWG has produced TIMES documents on sampling recommendations and quality assurance (Rees *et al.* 1991; Rees 2009, Rumohr 1990, 1999, 2009). The phytobenthos edition is just in its final stage (Kautsky 2010). This expertise enabled ICES to formulate advice to OSPAR and HELCOM and to revise and update the JAMP guidelines.

2.4.3 Availability of survey data

Two major databases combining a number of soft bottom macrozoobenthic species occurrence data have been created within the scope of EU Network of Excellence on Marine Biodiversity and Ecosystem Functioning (MarBEF) (Heip *et al.* 2009):

- 1) MacroBen integrated database of benthic invertebrates on European continental shelves; and
- 2) A database created within the scope of LargeNet responsive mode project.

Both datasets are going to be incorporated into the Ocean Biogeographic Information System (www.iobis.org) and will be thus accessible to scientists and open public.

MacroBen database has resulted from a large collaborative effort of MarBEF Theme 1 action that aimed to describe large-scale patterns of marine diversity (Vanden Berghe *et al.* 2009). The database integrated a total of 46 datasets provided by 24 European institutions. The database focussed only on soft bottom macrozoobenthos. The original datasets were collected within the scope of different research and monitoring programmes within the period of time ranging from 1937 to 2005. Geographically the database covers the northeast Atlantic and the North Sea, Norwegian Sea and Arctic Ocean, the Baltic Sea and the Mediterranean and Black Seas (Figure 2.3, Table 2.3.). It contains the quantitative and qualitative data on macrozoobenthic species distributions, based mostly on grab samples as well as some basic environmental parameters. The taxonomic names of species were validated against the European Register of Marine Species (ERMS). Altogether the database includes over 460 000 distribution records for over 7200 valid benthic taxa. The database has been used by teams of data providers to produce a number of papers addressing different aspects of large scale patterns of benthic diversity in European continental waters. Eight papers have been published within a special Theme Section in the journal *Marine Ecology Progress Series* (2009, volume 382).

LargeNet (Large-scale and long-term networking on the observation of Global Change and its impacts on Marine Biodiversity) was one of the smaller projects undertaken within the Theme 1 MarBEF activities (<http://www.marbef.org/projects/largenet/index.php>). The LargeNet database includes distribution records on both plankton and benthos and it covers three main habitats: 1) rocky shores, 2) soft bottoms and 3) pelagic environments (Vandepitte *et al.* in press). Altogether 67 datasets were provided by 19 institutions. 26 datasets were focused on soft bottom macrozoobenthos. These soft-bottom datasets came from the Arctic, Baltic Sea, North Sea and the Mediterranean (Figure 2.4; Table 2.4). The database includes a number of continuous time series datasets that together span a period of time from 1858 to 2007.

ICES ecoregions A and B (Greenland and Iceland Seas and Barents Sea) overlap with the geographical area covered by another large initiative aiming at large scale collection and integration of species diversity data – Arctic Ocean Diversity (ArcOD). ArcOD is a project within the Census of Marine Life programme. It aims to inventory biodiversity in the Arctic seas by use of compilation of existing data, taxonomic identifications of collected material, as well as new collections focusing on existing taxonomic and geographical gaps. The recent ArcOD initiative – a review of the pan-Arctic continental shelf benthic diversity led by Dieter Piepenburg (University of Kiel, Germany) and Philippe Archambault (Université du Québec à Rimouski, Canada) was successful in compiling a large dataset combining over 58,000 records of 2,788 macro- and megabenthic species. The database includes 106 stations covering the ICES ecoregion A (Greenland Sea, East and North Greenland) and 1,988 stations

sampled in the ICES ecoregion B (Barents Sea and White Sea). All species names were validated against WoRMS. The basic patterns of diversity on Arctic shelves have been analysed with use of the compiled database and will be described in a review paper being prepared by Dieter Piepenburg, Philippe Achambault and their co-authors (Piepenburg *et al.*, in preparation). The data will be available through the Ocean Biogeographic Information System (www.iobis.org) and the Global Biodiversity Information Facility (www.gbif.org).

In the ICES Ecoregion G benthic infauna diversity has been monitored within the context of several studies (taxonomic, ecological, assessment of human impacts, etc.) and is well documented (see OSPAR 2000, for the Quality Status Report for the North-East Atlantic, Bay of Biscay and Iberian Coast). In Portugal the assessment of the benthic infauna diversity has resulted mainly from small scale, short to medium-term studies, carried out on coastal and estuarine environments as well as coastal lagoons for communities' structural characterisation, ecological quality and environmental impact assessment purposes (Quintino, 1988; Neves *et al.*, 1991; Rodrigues, 1992; Gaudêncio & Guerra, 1994; Guerra & Gaudêncio, 1996; Desgarrado, *et al.*, 1997; Carvalho *et al.*, 2001; Costa *et al.*, 2003; Cunha & Ravara, 2003; Ferreira & Andrade, 2003; Cabeçadas *et al.*, 2004; Carvalho *et al.*, 2005; Canário *et al.*, 2007; Costa e Silva *et al.*, 2008; Gaudêncio & Cabral, 2007; Henriques *et al.*, 2008, 2009; Gaudêncio *et al.*, 2009, etc.). Since 2007 the surveys carried out by the IPIMAR benthos laboratory within the framework of the LIFE project Biomares, "Restoration and management of Biodiversity in the Marine Park Site Arrábida-Espichel" have been an important contribution to the monitoring of benthic infauna diversity in the Portuguese coastal area (Cunha *et al.*, 2010). The 2-year programme "Ecological Quality Assessment of Coastal and Transitional waters" under WFD (started in 2009 with sampling surveys from northern to southern Portugal) will also be a major contribution. Traditionally the mentioned studies include qualitative and quantitative species analysis and characterisation of grain size and organic matter content of the sediments.

As in the case of Portuguese waters, studies on the benthic infaunal diversity of Spanish waters arise mainly from small scale, short-medium term studies, carried out on coastal and estuarine environments for the characterisation of faunal communities, ecological quality and environmental impact assessment purposes (e.g. López-Jamar, 1978, 1986; López-Jamar & Mejuto, 1985, 1986, 1988; López-Jamar *et al.*, 1986, 1995, 1996b; Parra *et al.*, 2002; Palacio, 1996; Sánchez Mata *et al.*, 1993). More extensive studies in continental waters were carried out by López-Jamar & González (1987), López-Jamar *et al.* (1992) and Parra *et al.* (2002). Regarding long-term studies, there are some carried out by the Instituto Español de Oceanografía (IEO) in A Coruña (NW Spain); this project includes the study of the long-term variation of benthic infauna in two stations in A Coruña Bay (1982–present) (see López-Jamar *et al.*, 1995, 1996b; Parra *et al.*, 2008).

In addition to the above studies, there have been several other studies assessing the biodiversity and ecological status after two major oil spills in the area, the 'Aegean Sea' in 1992 and the 'Prestige' in 2002 (see López-Jamar *et al.*, 1996a; Parra, 2007; Parra *et al.*, 1994, 1995, 1997, 2005; Parra & López-Jamar, 1997; Mora *et al.*, 1996, 2003; Sánchez Mata, 1996; Garmendia, 1997; Gómez-Gesteira & Dauvin 2000 and 2005; Gómez-Gesteira, 2001).

2.4.4 Data quality issues

Standardisation of taxonomic nomenclature and validation of species names are major tasks included in each large scale study that involves merging several separate

datasets. A validation of species names in combined datasets includes correcting spelling errors and synonyms. It was shown in a case study performed for the LargeNet dataset that after the quality control and validation process reduced the number of unique taxon names by about 30 % (from the original number of over 6172 names to 4252 valid names). The European Register of Marine Species (ERMS), which has been incorporated into the World Register of Marine Species (WoRMS, <http://www.marinespecies.org/>), has become a most comprehensive and versatile tool to validate the species lists of the datasets.

There is a general consensus among Portuguese benthic ecologists about field and laboratory procedures generally following methods internationally agreed and recommended (ICES/BEWG reports; Holme & McIntyre, 1984; Eleftheriou & McIntyre, 2005). Apart from each laboratory internal quality control of the data there are no national quality control schemes or protocols. At the IPIMAR benthos laboratory all the taxonomic names are checked and standardised according to MARBEF/ERMS and WoRMS databases. The production of guidelines in order to standardize a number of procedures such as, for instance, species counting should be envisaged since it may help to improve data quality.

2.4.5 Taxonomic issues

Solving taxonomic problems belongs to pertinent activities when running large databases with species inventories. Since ICES is certainly not a taxonomic clearing house, mechanisms have to be found or maintained to gather a high and accepted taxonomical standard of the species names. The European Register of Marine Species has taxonomic specialists for each group to maintain scientific integrity and reliability. Since ERMS is also the basis for the ICES species inventory the same mechanisms applies for the stored species names. In addition, the normal procedures by using the help of museum-based taxonomists and other scientific experts are still in force. During the MarBEF project, ICES related scientists had the chance to also use the MarBEF taxonomic clearing house service which will hopefully be maintained in the planned MarBEF+ phase.

The reliability of infaunal species identifications is highly dependent on the state of preservation of the specimens and for that reason some of them may only be classified to higher taxonomic levels (e.g. fragmented polychaetes). The level of expertise of the personnel, the accuracy of essential laboratory equipment, such as microscopes and the availability of identification guides and publications on particular taxa as well as pictorial keys are also important issues to be addressed.

2.4.6 Capacity for ICES to include infauna within 'biodiversity' advice

Ongoing monitoring programmes for other faunal groups or research projects could be used to monitor benthic infauna. For example, in Portugal a significant part of the knowledge on benthic infaunal communities has been acquired in the scope of contracts for environmental impact studies established with several institutions (private/public), such as harbour authorities, building contractors, etc.

Although many of the national institutes have time-series data for benthic infauna, many of these sites are inshore and were selected to monitor the impacts of various human activities, and there are fewer offshore sites.

2.5 Benthic epifauna (including commercial shellfish)

2.5.1 Introduction

Benthic epifauna and megafauna includes those larger, motile invertebrates on the seafloor (e.g. crustaceans, molluscs and echinoderms) as well as the sessile taxa that are attached to hard substrata (e.g. sponges, cnidarians, bryozoans, ascidians) and is therefore a diverse ecological group. Such taxa are often sampled with small trawls or dredges (typically used on finer sediments) or by camera etc. on rocky grounds. For practical reasons of gear type, cephalopods are included within fish (Section 2.7).

2.5.2 Current ICES expertise

Epibenthic expertise potentially straddles several ICES Expert Groups. Some epibenthic taxa are taken as part of 'traditional' benthic infaunal studies, and as such may be addressed by the BEWG, and this EG has the expertise in invertebrate taxonomy and data analysis. In terms of on-going/potential data collection, many epibenthic species, especially the larger and/or faster moving species, can be better sampled with towed gears and so those EGs involved in the coordination of demersal fish surveys (e.g. WGBEAM, IBTSWG) may have practical knowledge about this ecological group, although some such surveys may only record information for commercial invertebrates. Some specific epifaunal habitats may be addressed by other expert groups (e.g. deep-water corals/sponges are addressed by WGDEC).

It should also be noted that several internationally coordinated groundfish surveys have been involved in national and EC-funded projects to undertake epibenthic sampling (e.g. Zühlke *et al.*, 2001; Callaway *et al.*, 2002; Ellis *et al.*, 2002) with a standardised gear (the steel 2 m beam trawl described by Jennings *et al.* (1999)), and such surveys could continue to provide useful platforms for epibenthic monitoring over wide spatial areas providing that resource were available.

With regards commercial invertebrates (other than cephalopods, which are addressed in Section 2.7), epibenthic crustacean expertise in ICES straddles several expert groups (e.g. Working Group on Biology and Life History of Crabs (WGCRAB), Working Group on Crangon Fisheries and Life Histories (WGCRAN), Study Group on *Nephrops* Surveys (SGNEPS)). Some internationally coordinated and national surveys for commercial crustaceans also provide platforms for sediment samples for macrofauna (e.g. Silva and Leotte, 2007; Leotte *et al.* 2005).

2.5.3 Availability of survey data

Beam trawl surveys have protocols for data collection for invertebrates (ICES 2009c), although the range of species for which data are collected is relatively limited. Some national surveys, such as French surveys of nursery areas, and various Spanish and Portuguese ground fish surveys, are recording (identifying, weighing and sometimes measuring) all the individuals caught within a haul, including non-target benthic epifauna (see ICES 2009b).

Various crustacean surveys (e.g. Portuguese crustacean surveys) and IBTS surveys submit data on commercial invertebrates, and some of these surveys also collect information on other invertebrates captured, although these data may not be submitted to the DATRAS database if the species are not measured. In the Portuguese *Nephrops* surveys conducted between 2005–2009, a total of 183 different taxa (including 45 crustaceans) were recorded.

In addition to trawl surveys, there are several underwater surveys for counting the density of *Nephtrops* burrows, and although some laboratories record observations on other fauna seen, it is unclear whether or not the taxonomic resolution is suitable for robust analyses of biodiversity.

2.5.4 Data quality issues

There are several important data quality issues with regards epifaunal data from trawl surveys.

- Although the net is generally 'shaken down' to ensure that all fish are passed down to the cod end and sampled, many invertebrates can 'stick' to the net or ground gear and not pass down to the cod end. This material may pass down the net on subsequent tows, and so species may be recorded erroneously for subsequent stations (unless there are obvious, observable differences in the appearance of fresh and 'old' material).
- Not all fish surveys have experienced and/or qualified invertebrate taxonomists on board, and so there is the possibility of mis-identifications. Data quality for conspicuous and easily recognisable taxa (which are often the dominant groups sampled) will be better than for rare and/or problematic taxa. Nevertheless, the 'human factor' (i.e. who on board processes the invertebrate catch) will likely be a very important factor affecting the number and range of species recorded, which obviously has major implications for any biodiversity studies.
- Those laboratories collecting information on the invertebrates caught may only process the catch on board and not retain material that cannot be identified at sea (e.g. due to a lack of resource for subsequent laboratory identification). Hence, some material may only be recorded at a higher taxonomic level, which is often the case with some of the sessile taxa (e.g. hydroids, sponges, ascidians, bryozoans).
- There can be differences into how rigorous the sorting process is. For example small catches can be sorted in full, whereas larger catches often need to be sub-sampled. This is particularly an issue for beam trawl surveys, where the biomass of the invertebrate by-catch can exceed the biomass of fish (Ellis *et al.*, 2000).
- Catch processing can also be affected by type of catch. For example, if there is a large catch of a sessile, habitat-forming species (e.g. a large mass of hydroids, *Flustra*, ascidians or horse mussels) then some of the cryptic species associated with such biogenic structures may be overlooked.

2.5.5 Taxonomic issues

Identification guides are available for most epibenthic taxa (e.g. Hayward and Ryland, 1990 and references cited therein). The qualities of the data that are collected during groundfish surveys are highly dependent on the expertise of people on board, and so some information may only be collected at a higher taxonomic level (e.g. sponges, hydroids, ascidians). Data for the more conspicuous and easy to identify 'megabenthos' (e.g. the larger brachyuran crabs, starfishes, gastropods) will likely to be of a better quality.

2.5.6 Capacity for ICES to include epifauna within 'biodiversity' advice

The ICES community is relatively well placed to provide advice on some commercial shellfish (e.g. *Nephrops*).

Existing beam trawl and otter trawl surveys have the capacity to provide information on the larger epibenthos and megabenthos, thus improving the knowledge on the functioning of marine ecosystems and the assessment of biodiversity (Brind'Amour *et al.* 2009). Nevertheless, improved standardisation and/or reporting of sampling protocols and limitations would be required before multi-survey spatial trends in diversity could be examined, or temporal trends analysed. Obviously these trawl gears are designed for demersal (and some pelagic) fish, and the catch efficiency will be low for many invertebrate species. However, in the case of some of the larger invertebrates, such towed gears may be one of the more effective sampling tools.

EC-funded projects did enable some groundfish surveys to undertake additional epifaunal sampling (with 2 m beam trawls), although not all nations have maintained this after the end of the projects. Groundfish surveys do offer a cost-effective way of undertaking such broadscale sampling, but resource is required to ensure appropriate scientific expertise on board and to allow the laboratory identification of some of the material.

2.6 Habitats and biotopes

2.6.1 Introduction

Several habitat features in the ICES area are of ecological and/or conservation importance and such habitats may be related to infaunal (Section 2.4) or epifaunal (Section 2.5) features.

Habitats are an essential part of biodiversity, and according to the Convention on Biological Diversity (CBD) a "Habitat" means the place or type of site where an organism or population naturally occurs. In the following account, the term 'habitat' should also encompass 'biotope' which is the environment in which a biological community occurs. Under EUNIS Habitat type is defined as "Plant and animal communities as the characterising elements of the biotic environment, together with abiotic factors operating together at a particular scale".

There are several important policy drivers for the monitoring of habitats and assessing habitat status.

2.6.1.1 Directive on the Conservation of natural habitats and of wild fauna and flora ("Habitats Directive", 92/43/EEC)

The EU Habitats Directive was adopted in 1992 as an EU response to the Berne Convention. It is one of the EU's two directives in relation to wildlife and nature conservation, the other being the Birds Directive. It aims to protect some 220 habitats (including some 20–30 marine habitats, see Table 2.5) listed in the directive's Annexes. These are the habitats which are considered to be of European interest, following criteria given in the directive. The most important management tool required by the directive is the installation of a network of Special Areas of Conservation, which together with the Special Protection Areas according to the Birds Directive is called Natura 2000.

2.6.1.2 Marine Strategy Framework Directive ('MSFD', 2008/56/EC)

As the "environmental pillar" of the European Maritime Policy, making habitats/biotopes a mandatory set of biodiversity elements to be assessed, the Marine Strategy Framework Directive (MSFD) came into force in June 2008. This directive establishes a framework within which "Member States shall take the necessary measures to achieve or maintain good environmental status in the marine environment by the year 2020 at the latest" (Art. 1).

Good environmental status is defined and measured, i.e., using a given set of environmental characteristics. Habitats are identified in descriptor 1 ("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"), and seabed habitats are also contained in descriptor 6 ("Sea-floor integrity is at a level that ensures that the structure and functions of the ecosystems are safeguarded and benthic ecosystems, in particular, are not adversely affected").

The habitat segment is given by the three broad categories in Annex III (Table 1): predominant habitats, special habitats and habitats of strategic importance. To specify the criteria and methodological standards based upon the Annexes, a Commission decision is expected to give comprehensive guidance (Art. 9). The EU COM has contracted ICES and JRC to develop recommendations on the criteria and standards in task groups assigned to the discrete descriptors of Annex I. Habitats/Biotopes were covered by Task Group 1 (Biodiversity) and Task Group 6 (Sea floor integrity). Task Group 1 has drafted suggestions on the treatment of habitats as outlined in Table 2.6.

2.6.1.3 HELCOM

At the joint ministerial meeting of the Helsinki and OSPAR Commissions (Bremen, 2003) it was decided to enhance efforts in identifying species and habitats that are threatened and/or declining or in immediate need of protection. Largely based on national red lists, HELCOM published the "HELCOM lists of threatened and/or declining species and biotopes/habitats in the Baltic Sea area" in 2007, and this contained 16 different subtidal habitat types (Table 2.7). The list aims to guide HELCOM and Contracting Parties in setting priorities for further work on the protection of marine biodiversity in the Baltic Sea.

2.6.1.4 OSPAR

In the Biological Diversity and Ecosystems Strategy of the Oslo-Paris Convention it was set out that OSPAR would assess which species and habitats need to be protected. Therefore, also the OSPAR Commission has developed a list of threatened and/or declining species and habitats (OSPAR 2008, see Table 2.8). The list is based on nominations of Contracting Parties and observers and on a selection process using agreed criteria ("Texel/Faial criteria"). The purpose of the list is to guide the OSPAR Commission in setting priorities for its further work on the conservation and protection of marine biodiversity.

2.6.2 Current ICES expertise

The current expertise of ICES on (offshore) marine habitats straddles several Working Groups, including the Benthic Ecology Working Group (BEWG), Working Group on Marine Habitat Mapping (WGMHM), Working Group on Ecosystem Effects of Fishing Activities (WGECO) and Working Group on Deep-water Ecology (WGDEC). Some other Working Groups may have knowledge on specific habitats, for example those Expert Groups addressing *Nephrops* may be able to assist with information on

'Sea-pen and burrowing megafauna communities'. Additionally, some habitat-forming species present on the continental shelf (e.g. *Modiolous*, *Sabellaria*) can be taken in groundfish and beam trawl surveys.

2.6.3 Availability of survey data

Whereas reporting according to the Habitats Directive Art. 17 in 2007 was still largely based on expert opinions, assessment in the next report (due in 2012) have to be based on new monitoring data. It is therefore mandatory for EU member states to monitor the relevant habitats in their areas, both inside and outside the Natura 2000 network of protected areas and to collect data on all parameters specified by the Commission (DocHab-04-03-03 rev3).

The Marine Strategy Framework Directive (Art. 11) prescribes the establishment and implementation of monitoring programmes to be based on the indicative lists of elements set out in Annex III (three categories of habitats), using the criteria and methodological standards as provided by the EU COM according to Art. 9 and by reference to the environmental targets established pursuant to Article 10. Reporting assessments based on monitoring results is due in 2018 and from then every six years.

For example, in Germany a harmonized national monitoring programme (BLMP) is being developed and implemented by the coastal 'Länder' (the federal states, for the 12 nm coastal zone) and the 'Bund' (Federal level, for the Exclusive Economic Zone (EEZ), 12–200 nm). Initially the habitats/biotopes segment of this programme targeted the habitat types to be assessed according to the EC Habitats Directive. However, the special habitat types listed by the regional sea conventions and the predominant habitats are currently being included according to the MSFD. In the EEZ, a respective monitoring programme started 2009.

2.6.3.1 Methods

A wide array of different methodological approaches to the monitoring of marine habitats/biotopes is available, including acoustic and visual remote sensing methods, grab sampling, dredging, beam trawling, diving and others. For any habitat type a specific set of adequate methods is to be selected and adapted to local and seasonal biotic and abiotic conditions.

2.6.3.2 Assessment

Guidance on the assessment of habitat status has been provided in the framework of EU directives (in particular according to the Habitats Directive: see DocHab-04-03-03 rev3). Member states implement these schemes in national assessment matrices specifying both parameters and thresholds.

Also within the framework of regional conventions assessment schemes have been provided, e.g. by OSPAR on habitats of the OSPAR list (see OSPAR "background documents").

2.6.4 Data quality issues

Fragile habitats are best sampled by non-destructive methods (e.g. visual census) and such methods will under-sample some or the smaller-bodied, cryptic or faster moving fauna, and so will only inform on a sub-set of the associated fauna.

2.6.5 Taxonomic and classification issues

In addition to the taxonomic issues described previously for infauna (Section 2.4.5) and epifauna (Section 2.5.5), issues of habitat classification are also an important issue.

One of several approaches to the classification of marine habitats is the EUNIS Habitat types classification, a pan-European system to facilitate a harmonised description and collection of data across Europe covering all types of habitats from terrestrial to freshwater and marine. The EUNIS classification is hierarchically organized and for marine habitats specified to four broad levels. At a fifth level and below, the component units are drawn from other classification systems and combine these in the common framework. For example:

A: Marine habitats

A5 : Sublittoral sediment

A5.1 : Sublittoral coarse sediment

A5.14 : Circalittoral coarse sediment

A5.141 : [*Pomatoceros triqueter*] with barnacles and bryozoan crusts on unstable circalittoral cobbles and pebbles

The EUNIS classification system is operable and largely used (e.g. in the Natura 2000 database) and is suggested to be used under MSFD descriptor 1.

2.6.6 Capacity for ICES to include habitats and biotopes within 'biodiversity' advice

The ICES community should be relatively well placed to provide advice on some of the specific habitats mentioned under OSPAR and HELCOM. For example, WGDEC can address some of the deep-water habitats.

2.7 Ichthyofauna and cephalopods

2.7.1 Introduction

Fish (including agnathans, chondrichthyans, sturgeons and teleosts) represent one of the more 'conspicuous' elements of marine biodiversity and includes species of high commercial importance (including for recreational fisheries), high conservation interest, as well as species of cultural significance. Cephalopods are normally taken in fish surveys and so are also considered in this section.

2.7.2 Current ICES expertise

ICES has extensive experience in fish-related issues (see Table 2.9), with this expertise distributed among the various survey coordination groups, the many stock assessment groups, the Working Group on Fish Ecology (WGFE), as well as some of the other expert groups that are more targeted towards specific taxa or management issues. The Working Group on Cephalopod Fisheries and Life History (WGCEPH) collates information on cephalopods. Some survey data for cephalopods are also collected during internationally coordinated surveys (e.g. WGIBTS).

2.7.3 Availability of survey data

Broadscale field data from trawl surveys are available for many parts of the ICES area (for example, Figure 2.5), including the Baltic Sea, North Sea, southern and western

waters and deep-water, with these surveys collecting data on the wider demersal and benthic-pelagic fish assemblage. Additionally, there are other EGs that may address ichthyoplankton surveys, acoustic surveys, or surveys for a particular species of interest.

In terms of trawl surveys, most data for internationally-coordinated surveys are available on DATRAS (or will be available in the near future). There will be some historical and/or nationally-funded trawl surveys that are not stored in DATRAS.

Much of these data are collected from surveys originally intended to investigate the distribution, relative abundance and biological characteristics of commercially important species (e.g. plaice and sole in beam trawl surveys; gadoids and herring in GOV trawl surveys). Hence the choice of gear and design of sampling grid may not be effective for all elements of the wider fish community.

There are numerous definitions of a (biological) 'community', which generally highlight that it is "A group of interdependent organisms living and interacting with each other and the environment in the same habitat". Towed gears may pass over multiple discrete habitats, and the various constituent species within any community may have very different catchabilities. Hence the catch composition of a trawl sample is not necessarily reflective of a community. Although a variety of fish will be sampled (and quantified and measured, so that species and size data are available), data represent the fish catch, which may be better assumed to equate with the 'fish assemblage', given that the term 'assemblage' may be defined as "the result of adequate sampling of all organisms of a specific category in a defined place", (see Magurran, 1988).

More specific fish 'communities' that are not sampled effectively in most ICES-coordinated otter and beam trawl surveys would include those that associate with rock and reef habitats (i.e. areas where trawl gears would tend to get damaged) and estuarine and inshore communities (where research-vessels cannot operate safely, but for which many national surveys exist). Although some trawls (e.g. GOV) have a high headline height and will catch some of the more common pelagic fish, these gears are not effective for sampling epipelagic species, large pelagic fish or, in deeper waters, meso- and bathy-pelagic species.

2.7.4 Data quality issues

There are important elements of data quality that should be noted with regards utilising survey data for 'biodiversity science':

- Catch rates will be affected by gear type

Although catch rates may be computed for all the fish species sampled, the catchability of each species may be slightly different. For example, fast-moving species may evade the net, smaller fish may pass through the meshes, and species closely associated with the seabed may pass under the ground gear. Although standard gears are used in the North Sea, the spatially extensive southern and western IBTS surveys currently use a variety of trawls (various forms of GOV trawl, boca trawl, Norwegian Campelen trawl, etc.), and as such there will be differences in how effectively each survey samples fish.

- Trawl catches only sample a fraction of the wider fish community

As noted above, not all fish species are sampled effectively. Hence, the occasional instances of, for example, larger sharks or estuarine/inshore species (e.g. grey mullet) in survey data sets will be very sporadic.

- Tows can sample several discrete habitats in a single tow

Towed gears may pass over multiple discrete habitats, and the various constituent species within any community may have very different catchabilities. Hence trawl samples should only be viewed as providing information on the fish assemblage as opposed to the 'community' (see above).

- Database errors

Many national laboratories use electronic data capture systems to record data, and so fish may simply be recorded by a code. There have been occasions where a particular fish has been reported, whereas in reality the species was not caught (or misidentified), but simply had a similar code. Obvious mistakes of this type can often be corrected during Quality Assurance procedures.

- Misidentifications

The reliability of species-identification can be a problem for several fish taxa (see Daan 2001; ICES 2007; ICES, 2009b and references cited therein), although survey-coordination groups are trying to impress the need for accurate species identification in the various surveys and to improve identification material where possible. Taxonomic issues are discussed in greater detail in Section 2.7.5.

- Changes/differences in sampling protocol regarding the taxonomic resolution

Many surveys were originally intended to inform on commercial fish species, and so non-target fish species have traditionally been of 'secondary' importance. In recent years, given the increased interest in using trawl survey data for wider ecosystem and biodiversity studies, there may have been subtle changes in data collection. For example, some surveys may have:

- Improved the accuracy of the species separation of morphologically similar species over the time series
 - Increased attention to including some of the smaller-bodied fish species (e.g. gobies and clingfish) in the catch data.
 - Recorded some fish at a higher taxonomic resolution at the beginning of the time series, and subsequently identified to species level
 - Used an inconsistent approach to using higher taxa. For example, some surveys not identifying sandeels to species may have used *Ammodytes* spp. and Ammodytidae interchangeably, whereas (in northern European waters) the former refers to two species, the latter to five.
 - Changes in the scientific crew and the expertise of the crew have also an effect on the number and consistency of species identified. The participation of experts or students dedicated to a particular group would also affect the identification of the species that varies from year to year if the expertise is not maintained throughout the time series.
- Distribution (over space and time) of the survey in relation to the distributions of fish species

The location of survey stations or stratification of trawl surveys tends to be targeted to the main commercial fish species. As such, some habitats can be under-represented

in trawl surveys. Additionally, larger research vessels may avoid shallower water areas, and so inshore grounds and sand bank habitats in coastal seas may be excluded from surveys for safety reasons.

- Tow speed

Different surveys have different protocols regarding trawling speed, this depends also on the main targeted species, since slower speeds are used for benthic or ground related species, whereas faster speeds are used when the main target are faster swimmers such as mackerel or horse mackerel. Therefore the differences in tow speed between surveys, and a long time series could also have an effect in the species sampled, and influence the biodiversity use of a survey data set.

- Tow duration

There has been much discussion between statisticians and survey managers regarding if it is better to have more stations of shorter tow duration, and the balance between how representative the samples are and the number of stations sampled. Within this issue, changes in tow duration have been implemented in some surveys to be able to increase the number of stations sampled (see Section 2.7.4.1). Nevertheless, in terms of biodiversity, reducing the tow duration could affect the catchability of faster swimming species, and also have an effect on the likelihood of catching rare or low abundant species, thus affecting the data for biodiversity studies in different ways.

- Season of the survey

Since in many cases surveys target to measure recruitment strength and its variations from year to year, they are usually carried out on the same season, typically autumn/early winter (3rd and 4th Quarters), although this design is consistent over time and allows comparisons along the time series, it also supposes a caveat when the data are used for biodiversity science, since some species may appear in the surveyed area only/mainly during a certain season. Additionally, any biodiversity metric that incorporates abundance data may be affected by strong year class strength. Benthic studies often treat recruitment events with care and how fish recruitment may effect biodiversity metrics needs to be considered.

- Quality Assurance

Several of the survey-coordination groups are now addressing issues of data quality, although not all data have been checked and corrected (where possible).

- Data filtering

To reduce the effect of all the data quality issues mentioned in this section when survey data sets are used to address biodiversity, usually some data filtering is performed. These filters are usually subjective and can be highly dependent on the knowledge on the survey and or the species; usually the better the knowledge on the survey and the species, the more adequate is the filtering, but at the same time it is important to carefully document the filtering done, including the species filtered and the reasons to filter them to allow the reproducibility and the comparability of the studies.

2.7.4.1 Case study illustrating the effects of tow durations on Portuguese fish community surveys

The sampling protocol of the Portuguese fish surveys was modified during the time series. The tow duration lasted 60 minutes during the first part of the series (1997–

1998, 2000–2001) and 30 minutes during the second part (2002, 2005–2007). We assessed the effects of the change in tow duration on fish community indices (Table 2.10) using a three-way nested analysis of variance (ANOVA), where the response variable is the index tested (Table 2.10) and the effects are the tow duration (2 levels), the sampling year nested within the tow duration (4 levels per year), and the geographic strata in which the sampling is conducted (12 levels). All strata were sampled every year. We also computed the accumulation curves for the two tow durations.

The results from the nested ANOVA for the density and the species richness indicated a significant interaction between the 30 and 60-minute tow duration and the strata in which the sampling was conducted (Table 2.10). Shannon's and Pielou's diversity indices showed a significant nested term (Year within Tow duration) and a significant single effect of the tow duration. The graphical examination of the interaction (Figure 2.6) and the single effect (Figure 2.7) indicated that the 30-minute tow duration displayed higher values for almost all the indices in all the strata. Comparison of the species accumulation curves (SAC, Figure 2.8), i.e. the cumulative number of species against a sampling effort (number of sites), indicated that the sampling effort for the two tow durations was sufficient to collect most of the species present as both SAC easily reached an asymptote. The asymptotic value was however higher for the 30 minutes in comparison to the 60 minutes tow duration. Overall, the comparison of the density, the diversity indices and the SAC suggested that the efficiency of the Portuguese trawl surveys was increased by reducing tow duration from 60 minutes to 30 minutes.

2.7.5 Taxonomic issues

There are over 1300 marine fish species known from European waters, including the Mediterranean Sea (Costello *et al.*, 2001, 2004), although there are regular additions to our ichthyofauna, both in terms of documented range extensions and/or occurrence of vagrants into the ICES area (e.g. Acosta *et al.*, 2008) as well as the descriptions of new species, especially from deeper waters (e.g. Andriashev, & Chernova, 1997; Nakaya & Stehmann, 1998; Fukui & Kitagawa, 2006; Iglesias *et al.*, 2004; Stehmann *et al.*, 2008). Currently, there is no comprehensive and up-to-date list of which fish species occur in the ICES area or specific eco-regions.

There is a relatively extensive literature for identifying European fishes (Box 1), and although many of the more common fish species are generally identified with a form of 'gestalt' recognition, dichotomous keys and other manuals are available for more problematic taxa.

It should be noted that species identification based on recognition, although quick and pragmatic, can lead to rare or unusual species, or species occurring outside their normal range, being overlooked if sea-going staff are not familiar with the existence of other species and how to separate these species. For example, although many surveys have recorded low numbers of the distinctive-looking garfish *Belone belone* (and in some earlier fish guides this is the only member of both its genus and family listed), there have been records of the closely related short-beak garfish *Belone svetovidovi* in the ICES area, which many sea-going staff will not be aware of and so this species may be overlooked, with all garfish simply recorded as *Belone belone*.

Although the majority of available data will be based on correct species identifications for the more common species, uniquely distinctive species and/or commercially important species (although there can still be more general database errors), there are

some taxa that may be difficult for the non-specialist to identify at sea, and the quality for data for such species is less certain.

IBTSWG identified the following fish taxa of being concern in some of the surveys, with problems varying across the region (e.g. only some regions have multiple species within the taxa), or only affecting some life-history stages (e.g. juveniles):

Lampreys (Petromyzontiformes); smooth-hounds (*Mustelus* spp.); skates Rajidae); shads (*Alosa* spp.); argentines (*Argentina* spp.); clingfishes (Gobiesocidae); sticklebacks (Gasterosteidae); seahorses and pipefish (Syngnathidae); redfish (*Sebastes* spp.); scorpion Fish (*Scorpaena* spp.); sea scorpions (Cottidae); horse mackerel (*Trachurus* spp., were several species occur); sea breams (Sparidae); wrasse (Labridae); grey mullets (Mugilidae); eelpouts (Zoarcidae); snake blennies (Stichaeidae); blennies (Blennidae); dragonets (*Callionymus*); sand eels (Ammodytidae); gobies (Gobidae); topknots (*Phrynorhombus*, *Zeugopterus*); scaldfish (*Arnoglossus* spp.); soles and tonguefishes (e.g. *Bathysolea*, *Dicloglossa*).

With regard cephalopods, this is a relatively small taxon, at least on the continental shelf of the ICES area, although the diversity is higher in deeper water, where there is less survey effort.

In general, the larger and more common species are usually easy to identify (e.g. the octopuses *Eledone cirrhosa* and *Octopus vulgaris*, cuttlefish *Sepia officinalis*). On the other hand the reliability of identification for some of the smaller cuttlefish species (*S. orbignyana*, *S. elegans*) is unclear. Bobtail squid (Sepiolidae) are more speciose and some members are subject to taxonomic revision. The data quality for these may not be good.

Although there are comparatively few species of squid on the continental shelf, there can be mis-identifications between *Loligo forbesi* and *Loligo vulgaris* (especially as suckers on the tentacle clubs are damaged easily) and between the young of both of these species with *Alloteuthis subulata*. The identification of ommastrephid squids may be variable. Species such as *Todarodes eblanae*, *Illex coindetti* and *Todarodes sagittatus* are clearly recognised by the differences in mantle and fins length and there are identification keys available.

2.7.6 Capacity for ICES to include fish and cephalopods within 'biodiversity' advice

2.7.6.1 Species diversity

The trawl data stored in the DATRAS database will be a valuable asset to looking at spatial patterns and temporal trends in biodiversity. However, such data will need to be used with a high degree of caution. Changes in gear, sampling methodology (e.g. tow duration) and catch processing (e.g. the treatment of non-target species) could have profound implications on any resultant biodiversity metric (or indicator).

When data quality checks have been completed, the ICES community will be able to use data from internationally-coordinated trawl surveys to inform on the general 'diversity' of the fish assemblages on the trawlable grounds of the continental shelf of ecoregions E, G, G and L. Data will be more limited for deep-water areas, and trawl data from national surveys should be available for other areas (e.g. parts of ecoregions A–D).

Nevertheless, it should be recognised that trawl surveys only sample a proportion of the fish species in the ICES area and some ecologically-important groups of fish may not be addressed by any resultant biodiversity indicator.

2.7.6.2 Genetic diversity

There are few coordinated programmes to examine long-term changes in the genetic diversity of marine fish, and most published studies have resulted from individual, taxa-specific studies, such as undertaken for cod (Hutchinson *et al.*, 2001, 2003) and thornback ray (Chevolot *et al.*, 2008). There have also been several studies for salmonids (e.g. Skaala *et al.*, 2006; Ribeiro *et al.*, 2008).

Many existing groundfish surveys have collected biological material for various genetic studies undertaken by the wider scientific community.

2.7.6.3 Threatened and declining species

Several species of fish are listed as 'Threatened and declining' by OSPAR, or are listed on national, EC and international legislation and Conventions (Table 2.11). If the purpose of a 'biodiversity indicator' is to ensure that there is no loss of biodiversity, then it should be noted that some of the existing trawl surveys may not be effective or suitable for examining many of the species listed.

Indeed, of the species listed in Table 2.11, only *Squalus acanthias*, *Raja clavata*, *Raja montagui*, *Gadus morhua* and, to a lesser extent, *Dipturus batis* are taken regularly in groundfish surveys on the continental shelf of the ICES area and some of the other species (e.g. *Centrophorus granulatus*, *Centrophorus squamosus*, *Centroscymnus coelolepis*, *Hoplostethus atlanticus*) are taken in deep-water surveys along the edge of the continental slope.

However, other species (e.g. *Lampetra fluviatilis*, *Petromyzon marinus*, *Anguilla anguilla*, *Alosa alosa*, *Alosa fallax*, *Coregonus lavaretus oxyrinchus*, *Salmo salar*, *Hippocampus ramulosus*, *Hippocampus hippocampus*) may only be captured occasionally and sporadically, and groundfish trawl surveys are not considered appropriate sampling techniques for these species due to their (often) inshore distribution and morphology (e.g. eel-like body shape, or being fast swimming pelagic fish, or being cryptic species that are often hidden in weed and hydroids and sometimes missed during catch sampling).

Some of the larger pelagic fish (e.g. *Cetorhinus maximus*, *Carcharodon carcharias*, *Lamna nasus*, *Mobula mobular*, *Thunnus thynnus*) are not sampled in existing surveys (although trawl surveys have caught individual *Lamna nasus*), and even some of the largest demersal species, which may have been locally found in parts of the ICES area at some point are now considered rare or extirpated from much of the former range, and so surveys have a low chance of catching these species (e.g. *Squatina squatina*, *Rostroraja alba*, *Acipenser sturio*, *Pristis* spp.)

Hence, if managers wish to use those trawl surveys coordinated by ICES to inform on 'fish biodiversity' and 'biodiversity loss', it may be that some of the species that may be of greatest conservation concern may not actually be present in the datasets.

Given that there are certain taxa of fish that may be more prone to biodiversity loss (e.g. diadromous fish, elasmobranchs), it may be possible that these groups of fish could be examined in more detail when examining biodiversity issues.

Box 1: Useful field guides for the identification of fish

Bauchot, M.L. and Pras, A. (1980). *Guide des poissons marins d'Europe*. Delachaux & Niestlé, 427 pp.

Corbera, J., Sabatés, A. and Garcia-Rubies A. (1998). *Peces de mar de la península Ibérica*. Planeta, Barcelona, 312 pp.

Louisy, P. (2006). *Guía de identificación de los peces marinos de Europa y del Mediterráneo*. Ediciones Omega, Barcelona, 458 pp.

Lozano Rey, L. (1928). *Fauna Ibérica: Peces*. Museo Nacional de Ciencias naturales, Madrid, 692 pp.

Lozano Rey, L. (1947). *Peces Ganoideos y Fisóstomos*. Memorias de la Real Academia de Ciencias, Madrid, 839 pp.

Lozano Rey, L. (1952a). *Peces Fisoclistos, Subserie Toracicos, Primera Parte (Ordenes Bericiformes, Zeiformes, Perciformes, Escorpeniformes y Balistiformes)*. Memorias de la Real Academia de Ciencias, Madrid, 378 pp.

Lozano Rey, L. (1952b). *Peces Fisoclistos, Subserie Toracicos, Segunda Parte (Ordenes Labriformes y Escombriformes)*. Memorias de la Real Academia de Ciencias, Madrid, 703 pp.

Lozano Rey, L. (1960). *Peces Fisoclistos, Subserie Toracicos (Ordenes Equeniformes y Gobiformes, Pediculados y Asimetricos)*. Ictiologia Iberica, Madrid, 613 pp.

Muus, B.J., Nielsen, J.G., Dahlstrøm, P.D. and Nyström, B.O. (1999). *Sea fish. Scandinavian Fishing Year Book*, 340 pp.

Nobre, A. (1935). *Vertebrados (mamíferos, reptis e peixes)*. Fauna marinha de Portugal 1, 574 pp.

Poll, M. (1947). *Poissons marins. Faune de Belgique*, 452pp.

Quéro, J.C., Porché, P. and Vayne, J.J. (2003). *Guide des poissons de l'Atlantique européen*. Delachaux & Niestlé, Paris, 465 pp.

Quéro, J.C. and Vayne, J.J. (1997). *Les poissons de mer des pêche françaises*. Delachaux & Niestlé, Laussane, 304 pp.

Redeke, H.C. (1941). *Fauna van Nederland X. Pisces (Cyclostomi-Euichthyes)*. A. W. Sijthoff's Uitgeversmaatschappij, Leiden, 331 pp.

Tortonese, E. (1956). *Leptocardia, Cyclostoma, Selachii*. Fauna d'Italia 2. Calderini, Bologna, 334 pp.

Tortonese, E. (1970). *Osteichthyes. Pesci ossei (parte 1)*. Fauna d'Italia 10. Calderini, Bologna, 565 pp.

Tortonese, E. (1975). *Osteichthyes. Pesci ossei (parte 2)*. Fauna d'Italia 11. Calderini, Bologna, 636 pp.

Wheeler, A. (1969). *The fishes of the British Isles and North-west Europe*. Michigan State University Press, 613 pp.

Wheeler, A. (1978). *Key to the Fishes of Northern Europe*. Frederick Warne, London, 380 pp.

Whitehead, P.J.P., Bauchot, M.-L., Hureau J.-C., Nielsen J., and Tortonese E. (eds) (1984-1986). *Fishes of the North-Eastern Atlantic and the Mediterranean*. 3 Vols, UNESCO, Paris 1473 pp.

2.8 Microalgae

2.8.1 Introduction

Microalgae, cyanobacteria and macroalgae are the primary producers of the ocean and play a major role in plankton ecology and ocean food webs. They are very relevant in terms of climate change, human health (water and seafood toxicity), the economy (through the effects on fisheries, coastal tourism, ballast water) and biotechnology (biofuels, bioactive molecules, nano-technology).

Microalgae are unicellular photosynthetic organisms occurring in both fresh water and marine environments. Many of them are planktonic, whereas others may live in sandy shores or as epiphytes attached to other organisms. Many microalgae have close relatives that are heterotrophic and are, together with other unicellular organisms, often termed protists or eukaryotic microbes. Phylogenetically, the microalgae as well as the protists are a very heterogeneous group. Estimates of extant algal species range from a conservative account at ca. 36 000 species (John & Maggs 1997) including ca. 10 000 macroalgal species (see below) to more than 10 millions (Norton *et al.* 1996). Species of microalgae new to science are continuously being described and the rate of discovery of new species from environmental samples is very high. The number of protists described to date is widely acknowledged to be only a fraction of the total diversity found in nature (Adl *et al.* 2007). The number of microalgae species present in the ICES region is for the same reason difficult to assess.

In terms of biomass the diatoms and the dinoflagellates are likely the two dominant groups in the North Atlantic. Both groups are well represented through the annual cycle, while the diatoms dominate during the spring bloom, the dinoflagellates tend to be more abundant in the summer and as autumn approaches. Several species of diatoms and dinoflagellates may form blooms and some of them are toxic (e.g. *Alexandrium tamarense*, *Dinophysis acuta*, *Pseudo-nitzschia multiseriata*). Another group of microalgae, the haptophytes (coccolithophyceans) are common inhabitants of the plankton and genera like *Phaeocystis* and *Emiliania* may cause extensive and in some cases harmful blooms.

As with most other marine regions, the North Atlantic is insufficiently sampled with respect to the microalgae, although the planktonic species of most diatoms, dinoflagellates and in some cases the haptophytes are fairly well known. Other habitats such as sandy shores and epiphytes of seaweeds, sea grass and animals have been less studied. Phytoplankton is normally collected using special water bottles or nets.

2.8.2 Current ICES Expertise

Within ICES, experts in the field of microalgae are currently active in the Working Group on Phytoplankton and Microbial Ecology (WGPME) and in the ICES–IOC Working Group on Harmful Bloom Dynamics (WGHABD). WGPME has been established recently (although there was the earlier Working Group on Phytoplankton Ecology), and WGHABD have produced annual reports since 1994 and these publications include reports from the member countries on harmful events in their regions.

2.8.3 Availability of survey data

Most European countries monitor their marine waters for harmful algae, but the extent of monitoring of other microalgae varies. Within the Water Framework Directive adopted by the EC, countries will have an obligation to monitor microalgae. The continuous plankton recorder (http://192.171.163.165/CPR_Survey.htm) has, since 1946, recorded phytoplankton collected by merchant ships in the North Atlantic, and this provides a good time series for studying temporal change for some microalgal groups (e.g. thecate dinoflagellates and diatoms) that fix well and can be identified in the light microscope. Several groups of microalgae are under-reported or over-looked due to either their small size and/or lack of distinguishing morphological characters.

2.8.4 Data quality issues

At the European level the International Standards Organization (ISO) has developed a standard for the quantification (routine analysis) of phytoplankton based on the

Utermöhl method, but the methods employed by the different countries and even within countries may vary.

2.8.5 Taxonomic issues

Several good microscopy guides for identifying and counting microalgae in the ICES area are available (e.g. Hällfors 2004; Thronsdén *et al.* 2005; Hoppenrath *et al.* 2009) and efforts are made within the scientific community to calibrate identification. Molecular methods are used increasingly (e.g. 454-pyrosequencing reveals diversity and barcoding for species identification) and many research programmes around the world are improving our knowledge of protist and microalgal diversity and the tools used for their identification.

2.8.6 Capacity for ICES to include microalgae within 'biodiversity' advice

The ICES community should be well placed to include some elements of the planktonic phytoplankton within biodiversity advice (e.g. through WGPME), although other microalgae, including non-planktonic groups, may be less well studied (although it is noted that BEWG have addressed phytobenthos in recent years, see Section 2.4.2).

2.9 Other elements of marine biodiversity

In addition to the taxa and broad 'ecological groups' (for the purposes of this section those taxa that tend to be taken in particular survey designs) described above, there are several other important taxa for which there was insufficient expertise at the meeting to address fully.

2.9.1 Macrophytes

Macrophytes include a variety of organisms, including macroalgae (Chlorophyceae, Phaeophyceae, Rhodophyceae), and flowering plants (e.g. *Zostera*, *Posidonia*, *Cymodoce*). The latter, as habitat features, are of conservation concern, as noted above (Section 2.6). Macroalgae would appear to have had less attention within the ICES community, despite the importance of such habitats for many species of marine invertebrates and fish, including juvenile gadoids (e.g. Gordon, 1983; Cote *et al.*, 2004; Norderhaug *et al.*, 2005; Konar & Hamilton, 2007; Laurel *et al.*, 2007; Perez-Matus *et al.*, 2007; Blight & Thompson, 2008).

2.9.2 Zooplankton

No experts on zooplankton could attend the meeting, although ICES has a well-established group that can ensure that zooplankton are addressed within biodiversity issues (Working Group on Zooplankton Ecology, WGZE).

Some of the larger gelatinous zooplankton (e.g. jellyfish) may also be taken in trawl surveys, but these specimens are generally damaged and so data are often not collected.

2.9.3 Seabirds

No experts on seabirds could attend the meeting, although ICES has a well-established group that can ensure that seabirds are addressed within biodiversity issues (Working Group on Seabird Ecology, WGSE).

2.9.4 Marine reptiles

Marine reptiles are not a major component of the marine fauna in the ICES area. Vagrants of five turtle species may occur: loggerhead turtle *Caretta caretta*, green sea turtle *Chelonia mydas*, hawksbill sea turtle *Eretmochelys imbricate*, Kemp's Ridley turtle *Lepidochelys kempii* and leatherback turtle *Dermochelys coriacea*. Two of these species (loggerhead and green turtle) have nesting beaches in the Mediterranean Sea (Márquez, 1990).

2.9.5 Marine mammals

No experts on marine mammals could attend the meeting, although ICES has a well-established group that can ensure that marine mammals are addressed within biodiversity issues (Working Group on Marine Mammal Ecology, WGMME).

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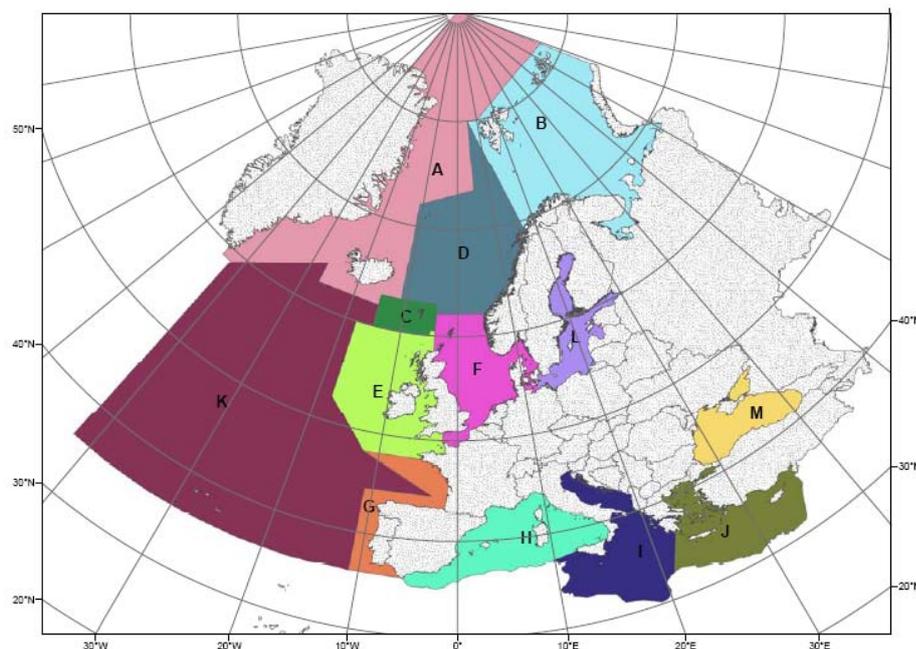


Figure 2.1. Ecoregions based on ICES Advice (A: Greenland and Iceland Seas; B: Barents Sea; C: Faroes; D: Norwegian Sea; E: Celtic Sea; F: North Sea; G: South European Atlantic Shelf; H: Western Mediterranean Sea; I: Adriatic-Ionian Seas; J: Aegean-Levantine Seas; K: Oceanic northeast Atlantic; L: Baltic Sea; M: Black Sea). The ICES area includes regions A-G, K and L.

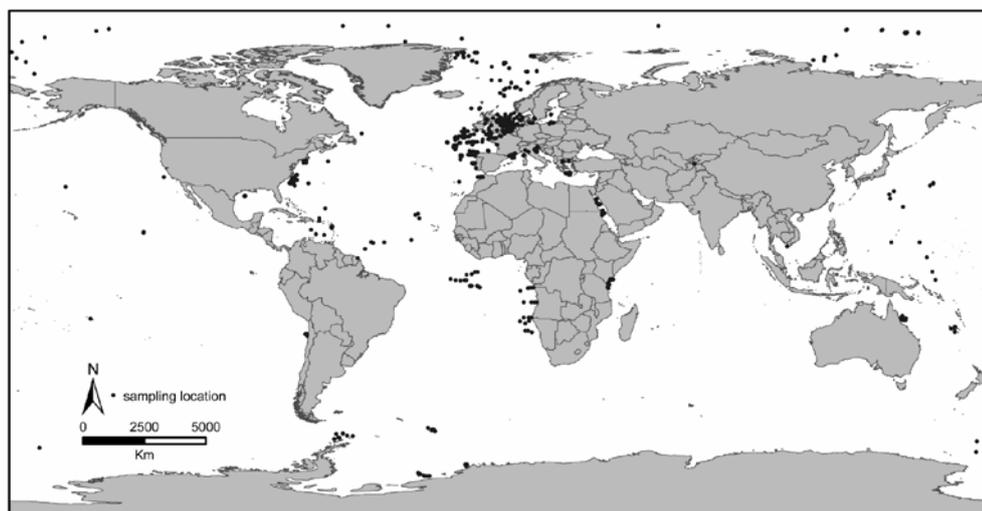


Figure 2.2. Meiofauna sampling locations available in the MANUELA integrated database (from Vandepitte *et al.* 2009).

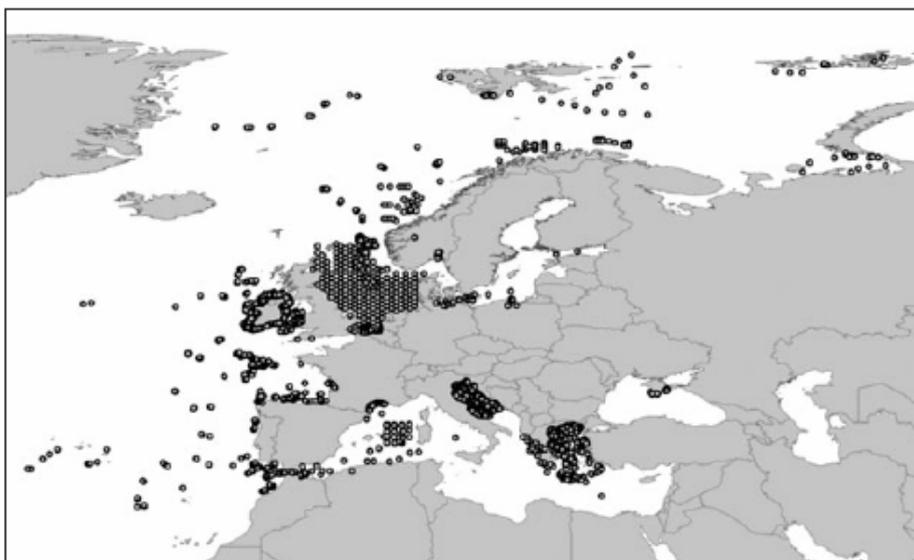


Figure 2.3. Soft bottom macrobenthic stations available in the MacroBen data base (from Vanden Berge *et al.* 2009).

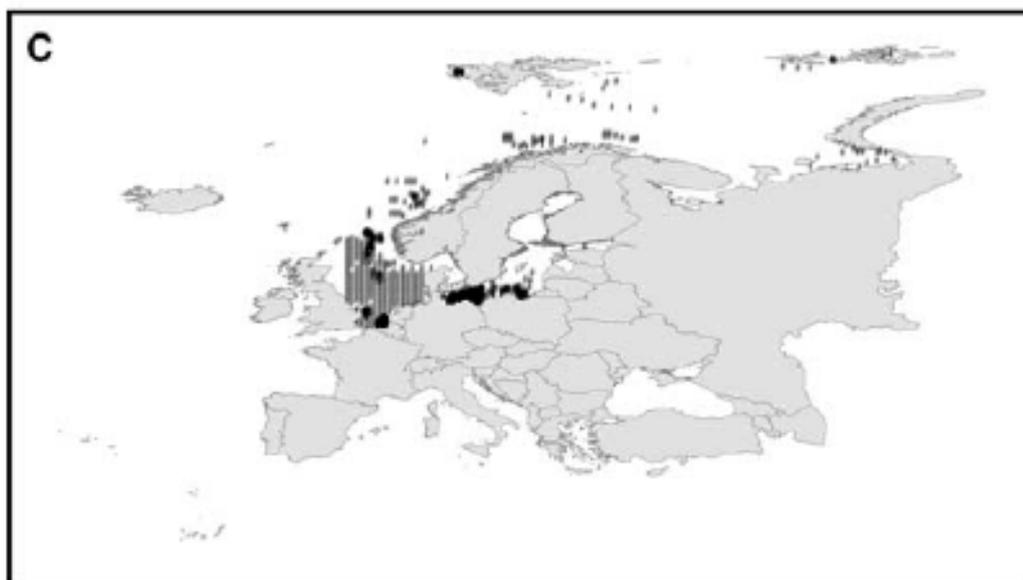


Figure 2.4. Soft bottom macrobenthic stations available in the LargeNet data base (from Vandepitte *et al.* in press).

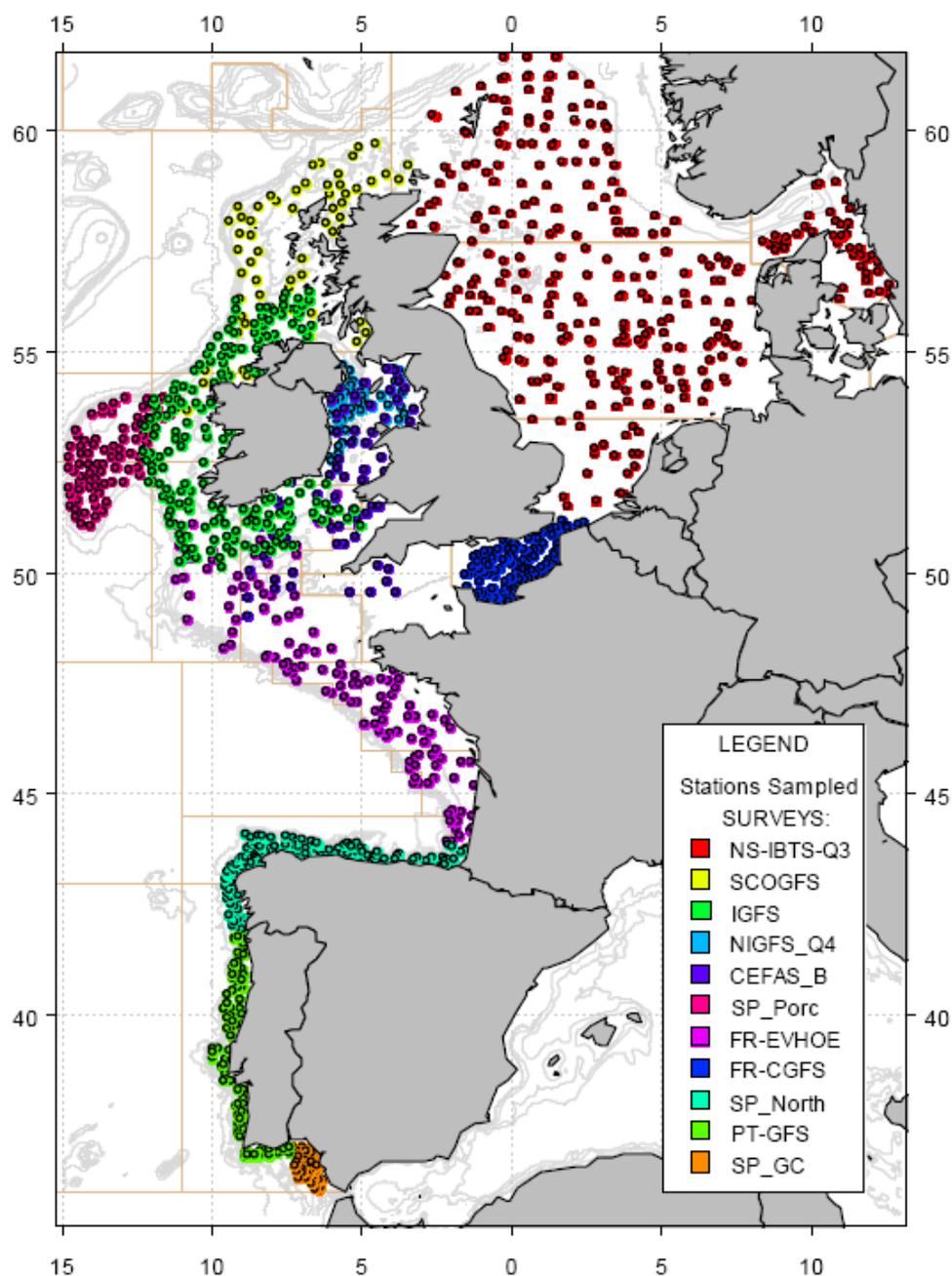


Figure 2.5. Distribution of IBTSWG-coordinated trawl surveys in the ICES area (Q3 and Q4 only). It should be noted that there is no standardised gear for the western and southern areas. (Source: ICES, 2009b). In addition to these surveys, there are also internationally coordinated beam trawl surveys, deep-water surveys and Baltic Sea surveys.

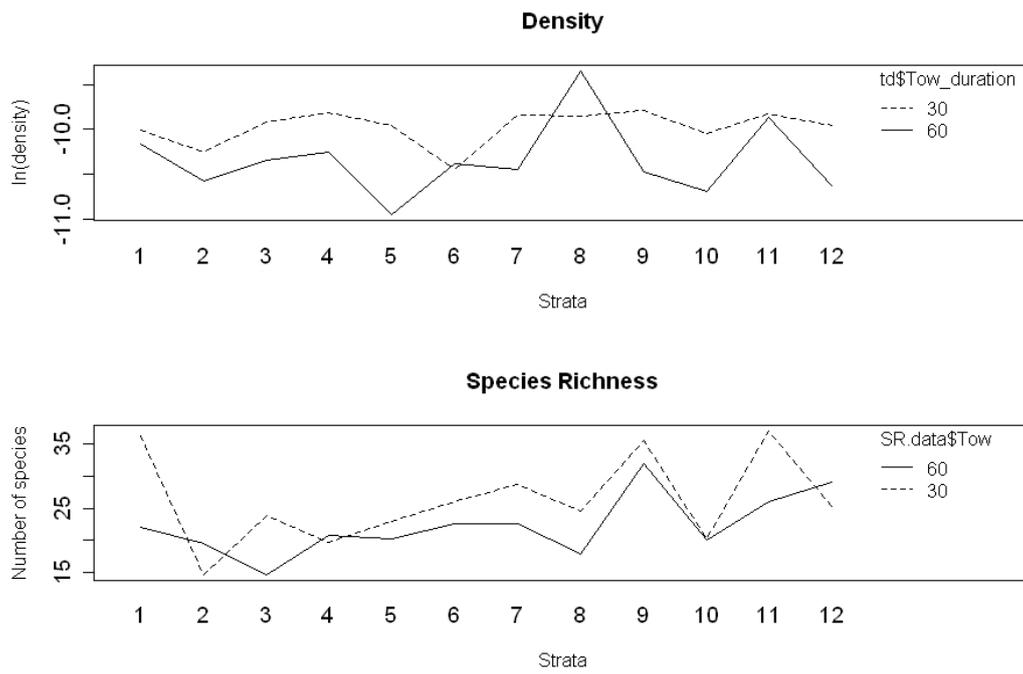


Figure 2.6. Interaction plot for the density and the species richness.

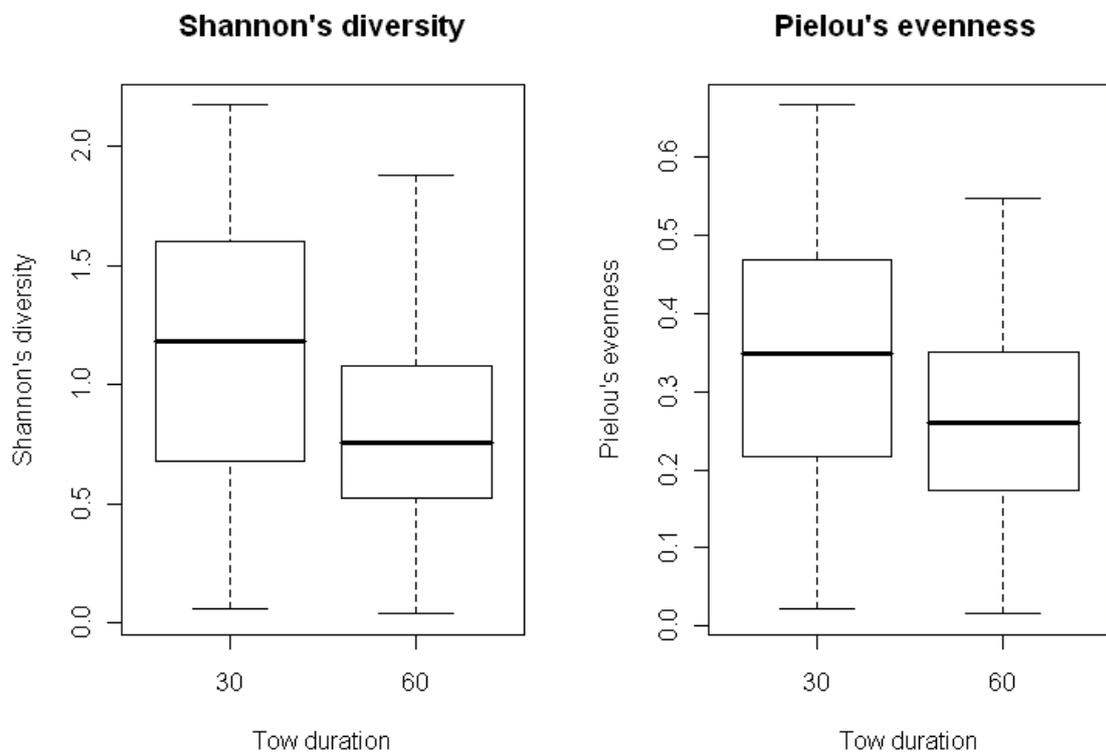


Figure 2.7. Mean diversity (Shannon's and Pielou's indices) estimated by tow duration

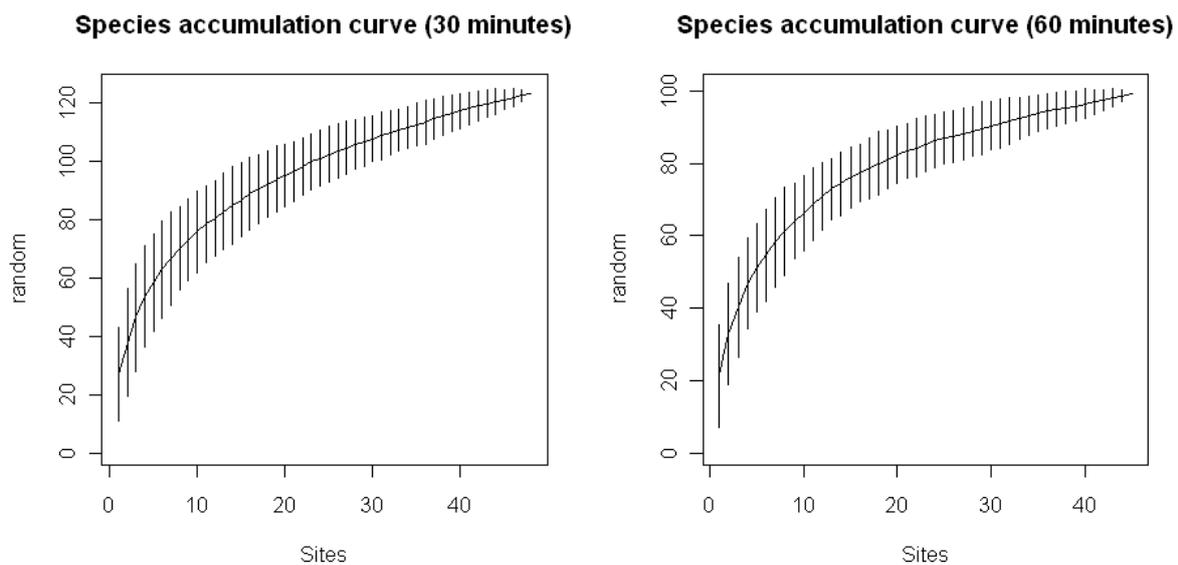


Figure 2.8. Species accumulation curves for the two tow durations.

Table 2.1. Ecological advantages and disadvantages associated with the use of meiofauna assemblages in applied sampling programmes (see Schratzberger *et al.* 2000 and references therein).

Advantages	Disadvantages
<p>Small size</p> <p>Meiofauna can be maintained in relatively small volumes of sediment. Therefore, intensive repeated sampling with minor disruption to the sampling site is possible because the sample size required is small. Assemblages are ideal for experiments on a small scale. Follow-up studies in the laboratory are possible under controlled and repeatable conditions.</p> <p>Ubiquitous distribution</p> <p>In particular nematodes occur in any environment that provides a source of organic carbon, under all climatic conditions and in habitats that vary from pristine to extremely polluted. They can colonise virtually every moist environment that can sustain metazoan life, and in marine sediments they usually constitute the most abundant taxon.</p> <p>High abundance and diversity</p> <p>A generally large number of individuals and species give a high intrinsic information value to each sample and ensure statistical validity of the data. High species diversity suggests a high degree of specificity in the choice of the environment.</p> <p>Short generation times</p> <p>Most species have short life-cycles (one to three months) so that changes in community structure and diversity can be observed in short-term studies.</p> <p>Direct benthic development, sessile habitat</p> <p>Meiofauna assemblages appear to be inherently more stable than those of larger organisms on a seasonal and year-to-year basis.</p>	<p>Taxonomic problems increase with smaller body size, whereas ecological knowledge decreases. A high-power microscope is required for species identification.</p> <p>Meiofaunal diversity and their community responses to environmental perturbations are not well documented, so there is not an extensive body of literature against which particular case histories can be evaluated.</p> <p>High abundance and diversity together with lack of taxonomic expertise can make the analysis of meiofauna a time-consuming and labour-intensive task.</p> <p>The spatial distribution of meiofauna is extremely patchy. Population density is affected by a variety of abiotic and biotic factors and consequently, meiofauna densities and diversity may fluctuate over distances of a few centimetres.</p> <p>Separating meiofauna from the sediment matrix requires a carefully controlled laboratory protocol.</p>

Table 2.2. MANUELA database: Number of datasets, number of sampled stations and number of distribution records in relation to the larger defined geographical areas. * These datasets contain information on several regions and can thus not be classified in one of the other regions (from Vandepitte *et al.* 2009).

	Datasets	Stations sampled	Distribution records
North-East Atlantic	29	215	50,755
North Sea	26	343	47,731
Mediterranean Sea	10	187	15,370
Arctic	6	33	2,689
Baltic	4	25	3,402
Antarctic	3	45	2,835
Indian Ocean	2	16	7,977
Global*	2	562	8,242
South Pacific	1	4	425

Table 2.3. MacroBen database: Number of datasets, number of sampled stations temporal range of data in different geographical areas. (after Vanden Berge *et al.* 2009).

Geographical area	Number of datasets	Number of stations	Temporal cover
North Sea and Northeast Atlantic Ocean	9	20171	1972–2005
Norwegian Sea and Arctic Ocean	8	333	1991–2003
Skagerrak and Baltic Sea	7	129	1980–2005
Mediterranean and the Black sea	20	1418	1979–1996

Table 2.4. LargeNet database: Number of datasets and temporal range of data in different geographic regions (from Vandepitte *et al.* in press).

Geographic region	Number of datasets	Temporal range
Arctic	7	1992–2006
Baltic Sea	13	1858–2007
North Sea	3	1874–2004
Mediterranean	1	2005
North Atlantic and Arctic	2	1990–2002

Table 2.5. Extract of Annex I of the Habitats Directive, highlighting marine and maritime habitats within the ICES area. Priority habitats are denoted *.

Natura 2000 Code	1. COASTAL AND HALOPHYTIC HABITATS
11	Open sea and tidal areas
1110	Sandbanks which are slightly covered by sea water all the time
1120	* Posidonia beds (<i>Posidonium oceanicae</i>)
1130	Estuaries
1140	Mudflats and sandflats not covered by seawater at low tide
1150	* Coastal lagoons
1160	Large shallow inlets and bays
1170	Reefs
1180	Submarine structures made by leaking gases
12	Sea cliffs and shingle or stony beaches
1210	Annual vegetation of drift lines
1220	Perennial vegetation of stony banks
1230	Vegetated sea cliffs of the Atlantic and Baltic Coasts
1240	Vegetated sea cliffs of the Mediterranean coasts with endemic <i>Limonium</i> spp.
1250	Vegetated sea cliffs with endemic flora of the Macaronesian coasts
13	Atlantic and continental salt marshes and salt meadows
1310	Salicornia and other annuals colonizing mud and sand
1320	Spartina swards (<i>Spartinion maritimae</i>)
1330	Atlantic salt meadows (<i>Glauco-Puccinellietalia maritimae</i>)
1340	* Inland salt meadows
14	Mediterranean and thermo-Atlantic salt marshes and salt meadows
1410	Mediterranean salt meadows (<i>Juncetalia maritimi</i>)
1420	Mediterranean and thermo-Atlantic halophilous scrubs (<i>Sarcocornetea fruticosi</i>)
1430	Halo-nitrophilous scrubs (<i>Pegano-Salsoletea</i>)
16	Boreal Baltic archipelago, coastal and landupheaval areas
1610	Baltic esker islands with sandy, rocky and shingle beach vegetation and sublittoral vegetation
1620	Boreal Baltic islets and small islands
1630	* Boreal Baltic coastal meadows
1640	Boreal Baltic sandy beaches with perennial vegetation
1650	Boreal Baltic narrow inlets

Table 2.6. Treatment of habitats, preliminary suggestions of ICES/JRC Task Group 1

Biodiversity component in Table 1 Annex III	Treatment for Descriptor 1
The predominant seabed and water column types	<p>To be treated together with their associated communities as combined assessments per habitat/community type.</p> <p>To be assessed according to the criteria for the attribute <u>habitat/community state</u>.</p> <p>To facilitate the comparability of monitoring results (Art. 11.2a) and consistency of assessment methodologies (Art. 8.3a), a consistent set of broadly-defined types (based on the EUNIS habitat classification) should be used across all regions and subregions.</p>
Special habitat types, especially those recognised or identified under Community legislation (the Habitats Directive and Birds Directive) or international conventions as being of special scientific or biodiversity interest	<p>To be assessed according to the criteria for the attribute <u>habitat/community state</u> (or <u>landscape state</u> where the listed feature is best considered as a marine landscape type).</p> <p>A set of relevant 'Listed' (special) types should be drawn up for each region/subregion.</p> <p>Their assessment may contribute in whole or in part to the assessments required for the predominant habitat types.</p> <p>They should be subject to a risk assessment process to ensure their assessment and monitoring is effective and efficient.</p>
Habitats in areas which by virtue of their characteristics, location or strategic importance merit a particular reference. This may include areas subject to intense or specific pressures or areas which merit a specific protection regime.	<p>Appropriate areas are expected to either be:</p> <ol style="list-style-type: none"> a) Areas subject to specific or multiple pressures and therefore addressed as part of the risk assessment approach for predominant and listed/special habitats, communities and species. b) Areas designated as marine protected areas (MPAs) or subject to other forms of protection, such as fishery closed areas. MPAs may provide suitable reference conditions to assist with assessment against targets in the wider region/subregion.

Table 2.8. Threatened and/or declining habitats in the OSPAR area (OSPAR 2008)

Habitat	OSPAR Regions where the habitat occurs	OSPAR Regions where such habitats are under threat and/or in decline
Carbonate mounds	I,V	V
Coral gardens	I,II,III,IV,V	All where they occur
<i>Cymodocea</i> meadows	IV	All where they occur
Deep-sea sponge aggregations	I,III,IV,V	All where they occur
Intertidal <i>Mytilus edulis</i> beds on mixed and sandy sediments	II,III	All where they occur
Intertidal mudflats	I,II,III,IV	All where they occur
Littoral chalk communities	II	All where they occur
<i>Lophelia pertusa</i> reefs	All	All where they occur
Maerl beds	All	III
<i>Modiolus modiolus</i> beds	All	All where they occur
Oceanic ridges with hydrothermal vents/fields	I,V	V
<i>Ostrea edulis</i> beds	II,III,IV	All where they occur
<i>Sabellaria spinulosa</i> reefs	All	II,III
Seamounts	I,IV,V	All where they occur
Sea-pen and burrowing megafauna communities	I,II,III,IV	II,III
<i>Zostera</i> beds	I,II,III,IV	All where they occur

Table 2.9. Summary of some of the main ICES Expert Groups in relation to various aspects of fish 'diversity' (note: this is not considered to be a definitive list).

Parent Committee	Name of Expert Group	Acronym	General remit with regards fish
ACOM	Arctic Fisheries Working Group	AFWG	Assess stocks of cod, haddock, saithe, Greenland halibut, capelin and redfish (<i>Sebastes</i> spp.) in ICES sub-areas I-II
ACOM	Herring Assessment Working Group for the Area South of 62°N	HAWG	Assess the various stocks of herring and sprat in ICES sub-areas III, IV, VI, VII
SSGESST	International Bottom Trawl Survey Working Group	IBTSWG	Coordinate groundfish surveys for sampling fish on the continental shelf of sub-areas IIIa, IV, VI-IX
ACOM	North-Western Working Group	NWWG	Assess stocks of cod, haddock, saithe, Greenland halibut, herring, capelin and redfish (<i>Sebastes</i> spp.) in ICES sub-areas V, XII and XIV
ACOM	Planning Group on Commercial Catches, Discards and Biological Sampling	PGCCDBS	Help with the coordination of biological sampling programmes etc.
SSGESST	Study Group on Standards in Ichthyoplankton Surveys	SGSIPS	Will review protocols for ichthyoplankton surveys
SSGSUE	Stock Identification Methods Working Group	SIMWG	Review issues of stock identity for (commercial) fish species, including genetic differences
SSGHIE	Working Group on the Application of Genetics in Fisheries and Mariculture	WGAGFM	Review effects of aquaculture, exploitation etc on the genetic structure of fish
ACOM	Working Group on Anchovy and Sardine	WGANSA	Assess stocks of sardine and anchovy in sub-areas VIII-IX
ACOM	Baltic Salmon and Trout Assessment Working Group	WGBAST	Assess salmon and trout in the Baltic
SSGESST	Working Group on Beam Trawl Surveys	WGBEAM	Coordinate beam trawl surveys in ICES sub-areas IV, VII and VIII
ACOM	Baltic Fisheries Assessment Working Group	WGBFAS	Assess various stocks (cod, sole, herring, sprat, flounder, plaice, dab, brill, turbot) in Baltic and/or Kattegat
SSGESST	Baltic International Fish Survey Working Group	WGBIFS	Coordinate acoustic and groundfish surveys of the Baltic
ACOM	Working Group for the Celtic Seas Ecoregion	WGCSE	Assess stocks of plaice, sole, cod, haddock, whiting in sub-areas VI-VII, megrim in IV, VI, and anglerfish in northern areas
ACOM	Working Group on the Biology and Assessment of Deep-Sea Fisheries Resources	WGDEEP	Assess the various deep-water stocks (e.g. greater argentine, beryx, orange roughy, black scabbardfish, red seabream, grenadiers etc.)
ACOM	The Joint EIFAC/ICES Working Group on Eels	WGEEL	Address issues on eels
ACOM	Working Group on Elasmobranch Fishes	WGEF	Assess stocks of elasmobranchs and address other issues relating to the taxonomic group
SSGESST	Working Group on North Sea Cod and Plaice Egg Surveys in the North Sea	WGEGBS	Will analyse and review the results of the 2004 and 2009 North Sea surveys for cod and plaice eggs

Parent Committee	Name of Expert Group	Acronym	General remit with regards fish
SSGHIE	Working Group on Environmental Interactions of Mariculture	WGEIM	Consider issues relating to impacts of cultured fish on wild populations
SSGSUE	Working Group on Fisheries-Induced Evolution	WGEVO	Review effects of exploitation on the life-history and genetic diversity of fish
SSGHIE	The Joint PICES/ICES Working Group on Forecasting Climate Change Impacts on Fish and Shellfish	WGFCCIFS	Examine the role of climate on the distribution and structure of fish communities
SSGEF	Working Group on Fish Ecology	WGFE	Address many aspects of fish ecology, including fish communities and diversity
ACOM	Working Group on the Assessment of Southern Shelf Stocks of Hake, Monk and Megrin	WGHMM	Assess southern and northern hake stocks, anglerfish in parts of sub-areas VII-IX, sole in the Bay of Biscay
SSGESST	Working Group for International Pelagic Surveys	WGIPS	Coordinate acoustic and larval surveys for herring and sprat in the North sea and adjacent waters
SSGESST	Working Group on Mackerel and Horse Mackerel Egg Surveys	WGMEGS	Coordinate ichthyoplankton surveys for mackerel and horse mackerel
SSGESST	Working Group on Northeast Atlantic Pelagic Ecosystem Surveys	WGNAPES	Evaluate various surveys for blue whiting, redfish, Norwegian spring-spawning herring, mackerel etc.
ACOM	Working Group on North Atlantic Salmon	WGNAS	Assess on North Atlantic salmon stocks
SSGESST	Working Group for North-east Atlantic Continental Slope Survey	WGNEACS	Coordinate surveys, including trawl surveys, in the deeper waters of the ICES area
ACOM	Working Group on Assessment of New MoU Species	WGNEW	Address sea bass, flounder, dab, lemon sole, brill, turbot, witch, red gurnard, tub gurnard, grey gurnard, striped red mullet and John dory
ACOM	Working Group on the Assessment of Demersal Stocks in the North Sea and Skagerrak	WGNSSK	Assess stocks of plaice, sole, cod, haddock, whiting, saithe, Norway pout and sandeels in sub-area IV and divisions IIIa and VIIId
SSGHIE	Working Group on Pathology and Diseases of Marine Organisms	WGPDMO	Consider issues relating to fish disease and impacts of cultured fish on wild populations
SSGEF	Working Group on the Science Requirements to Support Conservation, Restoration and Management of Diadromous Species	WGRECORDS	Consider issues relating to diadromous fishes (e.g. salmonids, eels, lampreys, shad)
SSGESST	Working Group on Redfish Surveys	WGRS	Plan the international trawl/acoustic survey of redfish in the Irminger Sea and adjacent waters
SSGEF	Working Group on Small Pelagic Fishes, their Ecosystems and Climate Impact	WGSPEC	Address various issues relating to small pelagic fishes such as herring, sardine, anchovy, sprat and capelin
ACOM	Working Group on Widely Distributed Stocks	WGWIDE	Assess the various wide-ranging stocks of horse mackerel, herring, mackerel and blue whiting

Table 2.10. Results from the three-way nested ANOVA estimating the effect of tow duration, the year nested in the tow duration, and the geographic strata in which the sampling was conducted. Df: Degree of freedom, SS: Sum of squares, MS: Mean square, p: Probability.

	Df	SS	MS	F value	p
In(Density)					
Tow duration	1	172.7	172.7	31.3147	<0.0001
Strata	11	155	14.1	2.5559	0.0032
Tow duration(Year)	6	107.7	17.9	3.2553	0.0034
Tow duration * strata	11	142.9	13	2.3569	0.0068
Residuals	3971	21893.8	5.5		
Species richness					
Tow duration	1	360.6	360.6	13.5026	0.0005
Strata	11	2398.71	218.06	8.1654	<0.0001
Tow duration(Year)	6	796.78	132.8	4.9726	0.0003
Tow duration * strata	11	737.76	67.07	2.5114	0.0109
Residuals	63	1682.47	26.71		
Shannon's index					
Tow duration	1	3.0462	3.0462	15.9974	0.0002
Strata	11	4.7703	0.4337	2.2774	0.0206
Tow duration(Year)	6	3.5356	0.5893	3.0946	0.0102
Tow duration * strata	11	3.5253	0.3205	1.6831	0.0980
Residuals	63	11.9963	0.1904		
Pielou's index					
Tow duration	1	0.22065	0.22065	12.15	0.0009
Strata	11	0.37643	0.03422	1.8844	0.0585
Tow duration(Year)	6	0.27333	0.04555	2.5085	0.0306
Tow duration * strata	11	0.34956	0.03178	1.7499	0.0827
Residuals	63	1.14408	0.01816		

Table 2.11. Conservation designations for marine fishes in the ICES area

Family	Scientific name	Common name	Bern Convention	OSPAR listed	Habitats Directive	CITES	Nominal range in ICES area
Petromyzontidae	<i>Lampetra fluviatilis</i>	River lamprey			Annex II		Occurs in coastal waters of ICES areas
Petromyzontidae	<i>Petromyzon marinus</i>	Sea lamprey		Yes	Annex II		Widespread in ICES area
Squalidae	<i>Squalus acanthias</i>	Spurdog		Yes			Widespread in ICES area
Centrophoridae	<i>Centrophorus granulosus</i>	Gulper shark		Yes			In deep-water of ICES area
Centrophoridae	<i>Centrophorus squamosus</i>	Leafscale gulper shark		Yes			In deep-water of ICES area
Somnositidae	<i>Centroscymnus coelolepis</i>	Portuguese dogfish		Yes			In deep-water of ICES area
Squatinae	<i>Squatina squatina</i>	Angel shark		Yes			Widespread in ICES area
Cetorhinidae	<i>Cetorhinus maximus</i>	Basking shark	Appendix II (Mediterranean)	Yes		Appendix II	Widespread in ICES area
Lamnidae	<i>Carcharodon carcharias</i>	White shark	Appendix II (Mediterranean)			Appendix II	Occurs in ICES area
Lamnidae	<i>Lamna nasus</i>	Porbeagle		Yes			Widespread in ICES area
Pristidae	<i>Pristis pectinata</i>	Smalltooth sawfish				Appendix I	Northern limits may extend to ICES area
Pristidae	<i>Pristis pristis</i>	Common sawfish				Appendix I	Northern limits may extend to ICES area
Rajidae	<i>Dipturus batis</i>	Common Skate		Yes			Widespread in ICES area
Rajidae	<i>Raja clavata</i>	Thornback ray		Yes (II only)			Widespread in ICES area
Rajidae	<i>Raja montagui</i>	Spotted Ray		Yes			Widespread in ICES area
Rajidae	<i>Rostroraja alba</i>	White skate		Yes			Widespread in ICES area
Mobulidae	<i>Mobula mobular</i>	Devil ray	Appendix II (Mediterranean)				Northern limits may extend to

Family	Scientific name	Common name	Bern Convention	OSPAR listed	Habitats Directive	CITES	Nominal range in ICES area
			terranean)				ICES area
Acipenseridae	<i>Acipenser sturio</i>	Atlantic sturgeon	Appendix II	Yes	Annex II, IV	Appendix I	Occurs in coastal waters of ICES areas
Acipenseridae	<i>Acipenser spp.</i>	Sturgeons				Appendix II	A. oxyrinchus reported from ICES area, vagrants from NW Atlantic?
Anguillidae	<i>Anguilla anguilla</i>	European eel		Yes		Appendix II	Widespread in ICES area
Clupeidae	<i>Alosa alosa</i>	Allis shad		Yes	Annex II		Widespread in ICES area
Clupeidae	<i>Alosa fallax</i>	Twaite shad			Annex II		Widespread in ICES area
Salmonidae	<i>Coregonus lavaretus oxyrinchus</i>	Houting		Yes	Annex II, IV (some populations only)		Widespread in northern areas and Baltic. Concern over North Sea populations
Salmonidae	<i>Salmo salar</i>	Salmon		Yes	Annex II (fresh-water only)		Widespread in ICES area
Gadidae	<i>Gadus morhua</i>	Cod		Yes (II, III only)			Widespread in ICES area
Trachichthyidae	<i>Hoplostethus atlanticus</i>	Orange roughy		Yes			In deep-water of ICES area
Syngnathidae	<i>Hippocampus ramulosus</i>	Sea-horse	Appendix II (Mediterranean)	Yes		Appendix II	Outside ICES area
Syngnathidae	<i>Hippocampus hippocampus</i>	Short-snouted sea-horse	Appendix II (Mediterranean)	Yes		Appendix II	Outside ICES area
Scombridae	<i>Thunnus thynnus</i>	Bluefin tuna		Yes			Oceanic species occurring in ICES area

3 Macroecological theories and rules of importance to understanding biodiversity

3.1 Biodiversity: theories and implications

This section gives a brief overview of some theories on ecological and evolutionary processes causing and regulating the number of species, and points to some implications of this for the development of indicators and management. This section is not meant to be an exhaustive review of all the theories of biodiversity, for this a number of good books and websites exist.

3.1.1 Processes regulating the number of species.

The processes controlling the development and maintenance of the number of species in time within a more or less geographically distinct area can be reduced to *Invasion*, *Extinction* and *Speciation*. While there would be little debate that these processes determine biodiversity, the degree of dissent between different theories is largely on the functional forms and the relevant importance of these processes. While these debates may seem academic, there seems to be a consensus that different ecological communities are regulated by different processes, though there are attempts to pose unifying theories (see for example Loreau 2010).

Theories of biodiversity have focused on different aspects of these processes, and have had different interpretations on the functional forms of these components of biodiversity change. Classically, island biogeography (Simberloff and Wilson 1969, MacArthur and Wilson 1967) has focussed on the processes of *invasion* and *extinction* as a function of island size and distance between islands. This has been influential in the development of species-area curves and to a certain extent in the debate on few large versus many small reserves in conservation planning.

More recently, niche dependant theories of biodiversity (Lotka 1932, MacArthur 1972, Tilman 1982) have focused on how coexistence can be promoted through niche differentiation in time and/or space: allowing for the partitioning of limiting resources (Tilman 1982, Huisman & Weissing 1999, de Roos *et al.* 2008) thus ameliorating *extinction* processes through so called “stabilizing” mechanisms. Competition-colonisation theories (Levins & Culver 1971; Tilman 1994, and others), emphasize the role of space, and explain how trade-offs between competitive and colonizing abilities regulate coexistence. Here increased *invasion* capacity (colonization) is coupled to a decreased capacity to promote *extinctions* (i.e. a decreased competitive ability).

Alternatively, so called “equalizing” processes may prevent competitive exclusion (*extinction*) by promoting the convergence of functional traits. This effectively reduces exclusion processes to a random drift of equivalent species in time, in so called neutral theory of biodiversity (Hubbell 2001). In this theory all species are alike and *invasions* are taken to be equivalent to *speciation*. Niche and neutral theories may both work concurrently to produce species clumps on trait axes (Scheffer and van Nes 2006) as recently demonstrated for phytoplankton communities (Vergnon *et al.* 2009).

In the intermediate disturbance hypothesis: (Grime 1973; Horn 1975; Connell 1978) biodiversity is proposed to be highest when disturbance, both with respect to intensity and frequency, is between low and high. Connell (1978) considered two causes for low diversity at either end of a disturbance continuum.

- i) The competitor that is most efficient in exploiting limited resources and/or the most effective in integrating with other species will eliminate

the rest. In other words, at low levels of disturbance, competitive interactions prevail, thus promoting competitive exclusions,

- ii) (Even if all species were equal in competitive ability, the one that is most resistant to damage or death will fill much of the space, i.e. at high levels of disturbance so-called stress tolerant species dominate.

Interactions between species may also regulate niche dependant processes that also lead to *speciation*. Here mutations in traits governing the functional ability of organisms to either compete with or predate on other organisms. Broadly speaking, allopatric *speciation* relies on evolution within geographically separate areas and consequent reintroduction (Rosenzweig 1995), and thus comes into play through the *invasion* component of the equation, while sympatric speciation relies on the niche-based interactions, such as competition, driving the specialization of organisms and the divergence of trait values as a consequence of eco-evolutionary interactions (see for example Dieckmann & Doebeli 1999; Louille & Loreau 2005; Fussmann *et al.* 2007), this development in trait values may also lead to *extinction* of species along the way.

3.1.2 Some implications

What processes determine and maintain biodiversity may strongly vary between communities and be related to the relevant spatial and time scales. Thus some communities may be characterised by species with fast generation times and thus a tendency for speciation to play/have played a dominant role in community formation. For example, in microbial communities speciation may be fast enough to play an important role in the natural structure and functioning of the community, and confer a natural turnover in diversity.

In other cases, where disturbances are a strong component in the regulation of biodiversity within communities, hump shaped relationships with disturbance intensity are to be expected, and non-intuitive increases in disturbance may lead to higher diversity. There is already strong evidence that evolutionary processes may play a role on human time scales, and that the implications of this evolutionary change have real consequences for natural populations (Jorgenson *et al.* 2007). The ramifications of this for food web processes and structure and functioning of ecological communities are still relatively unexplored.

3.2 Species and stock issues

The assessment of biodiversity in general is based on knowledge in systematics using species as the basic taxonomic unit for classification. Overall, species can be considered to be a fundamental category of biological organisation contributing to the taxonomic framework used in all branches of biology. In evolutionary theory the most commonly used species concept is the *Biological Species Concept* (BSC). It is largely influenced by ideas of Ernst Mayr (Mayr 1942) and states that a species consists of populations of organisms that can reproduce with one another and that are reproductively isolated from other such populations (de Queiroz 2005). Both reproductive isolation and hybridization barriers prevent production of offspring or ensure infertility. However, the most commonly *applied* concept in practice has always been the morphological species concept according to which species are described on the basis of morphological differences. Most zoologists' and botanists' work in species taxonomy is based on this concept, especially in terms of field identification.

Different mechanisms of reproductive isolation are genetically controlled. These can evolve in species with overlapping geographic distributions or as adaptive divergence leading to speciation. The mechanisms described ensure the integrity of a species over time by reduction or inhibition of gene flow between different species, thus allowing the conservation of each species characteristics. However, the species concept is still under debate including new conceptual ideas towards a rank-free taxonomy (Mishler 1999) or the description of genotypic clusters based on genetic divergence (Hendry *et al.* 2000). This is partly supported by the limitation of the BSC to describe new species so that other concepts like the Phylogenetic Species Concept (PSC) were designed. Within this concept, species are defined as populations of organisms that have a high level of genetic similarity. This might have been caused by adaptation to the same niche, and the transfer of genetic material from one individual to another. Understanding genetic diversity and how it is distributed has important conservation management applications, both for the resolution of taxonomic uncertainties surrounding species complexes and also for the identification of genetically distinct populations and the designation of reserves that retain maximum genetic diversity (Moritz, 1994).

The picture is more complex for microorganisms. As outlined in Section 2.2, genotype and phenotype characteristics are used to define new microbial species. Especially for prokaryotes an exchange of genomic information between organisms can occur via mechanisms like lateral gene transfer or homologous recombination. While lateral gene transfer can take place between different species homologous recombination is restricted to closely related organisms. For in-depth information, please see the review of Achtmann & Wagner (2008). Furthermore, lateral gene transfer can also occur with eukaryotes though far less frequently.

Although little studied in wild marine fishes (but see Pyle & Randall 1994; Roques *et al.* 2001; Hobbs *et al.* 2009), hybridization can occur between some pleuronectids. It is also known to occur within a variety of diadromous taxa, including salmonids, sturgeons and shads (e.g. Maitland & Lyle 2005; Alexandrino *et al.* 2006).

In addition to species-level information, many commercial fish and shellfish are traditionally assessed at the stock level, and other species may be considered at a regional/population level. The stock unit is usually described by the extent to which the effects of exploitation in a particular fishery are recognisable through a species' population (see Harden-Jones, 1968; Pawson & Jennings, 1996) and, ideally, this management unit should also have a high degree of biological integrity (i.e. it is a sub-population in which growth, recruitment and mortality are the main factors regulating population dynamics, with immigration and emigration of negligible importance). The stock concept is a potentially important issue in relation to biodiversity science. For example:

- Are the design and accuracy of current biological sampling programmes suitable to inform on the distribution and abundance of the stock?
- Is it an effective way of incorporating some degree of 'genetic diversity'?
- There may also be a need to better resolve the identification of 'threatened and declining species' (which is often species or species/area based) with the actual stock/population units monitored and assessed.

3.3 Succession, community regulation and 'baseline' conditions

Natural systems may reach higher stages of complexity until a level of saturation or the 'climax stage' is reached. There is some debate on the climax stage, but succes-

sional process and species turnover are less controversial. Most studies on succession and 'climax communities' in the marine environment have been on hard ground communities, often in inshore environments (e.g. Vance 1988), and they are less well understood on soft bottoms (but see Flach 2003).

Successional processes can be observed in many kinds of communities/assemblages and are governed by external factors such as larval supply, temperature, degree of disturbance in the system etc. Successional processes can be observed after physical disturbances (e.g. ice berg scouring, trawling, dredging operations, dredged spoil disposal), secondary sedimentation, chemical disturbance (e.g. sewage impact, point source pollution, oil spillages), and anoxic / oxygen deficient periods. Succession normally begins at defaunated sites with the early settlement of short-lived opportunistic species (so-called *r*-selected species) that can cover open substrates rapidly and in high numbers. These are followed by intermediate species which have higher demands on their environment and finally by *k*-selected species with low numbers of offspring, high environmental demands and normally longer life cycles.

Species grow with their prey size, so that always the optimum spectrum of food size is available for predators. We can observe successional processes in environmental gradients such as classically described by Pearson & Rosenberg (1978). Here we can observe them both in a spatial gradient but also in a time gradient, when we sample over an appropriate time sequence. Succession may not always reach theoretical climax but may stop earlier at intermediate stages due to environmental stressors and conditions. This intermediate stage is also regulated by predatory processes, breaks in larval supply and generally governed by environmental parameters such as temperature, salinity, oxygen, light and many other factors.

Arntz & Rumohr (1982, 1986) studied macrobenthic colonization and succession, and the importance of seasonal variation in temperate latitudes in the Western Baltic and documented especially fluctuations of benthic macrofauna during succession and in an established community. Succession can be quite fast (1–3 years) but can also be quite long in extreme environments, such as in polar or deep-sea environments. Successional processes can be either spatially widespread but also locally restricted. Examples are succession events after point source discharges of sewage or pollutants or succession of users on whale carcasses (so-called whale falls). This process may last for several years until all available resources are used up by a successional sequence of specialized users.

Baseline studies are, ideally, a pre-requisite to characterize marine ecosystems according to their specific environmental features and communities, including the assessment of spatial and temporal changes of features such as the number of species, their abundances, biomass, breeding and migratory patterns etc. Hence, they help to analyze relationships with a range of natural and human-induced variability, and how the systems might be changed by different impacts. For biodiversity conservation purposes, baseline studies should be regarded as one of the first steps in providing the necessary information on reference conditions in order to assess changes, set targets and ultimately to develop management measures and decisions to prevent the loss of biodiversity. However, it must also be recognized that after many decades of human activities in the marine environment, especially on the continental shelf, it may not be possible to quantify or identify what 'baseline' conditions should be.

3.4 Habitat complexity

Habitat complexity is closely related to the ecological niche theory in the sense that it provides a variety of potential ecological niches that may be filled by different species, thereby favouring the coexistence of these species (MacArthur and MacArthur 1961). For instance, littoral habitats containing woody debris and leaf litter might support higher primary and secondary productivity, thereby supplying fish with multiple opportunities for foraging strategies (Willis *et al.* 2005). Several aquatic studies appreciated significant associations between measures of diversity (e.g. alpha diversity: species richness, assemblage diversity) and measures of habitat complexity (Hunter and Sayer 2009). However, as the ecological niche is composed of many dimensions direct estimation of community niche relationships is often a daunting challenge even in species-poor systems.

Thrush *et al.* (2001) demonstrated that broad scale anthropogenic disturbances that reduce the density of epifauna and homogenise surficial sediments can have important consequences for seafloor biodiversity. They investigated the habitat structure and macrofaunal diversity of relatively simple soft-sediment habitats over a number of spatial scales (cm to km) to identify the role of habitat structure in influencing macrobenthic diversity and to assess the validity of using habitat structure as a surrogate measure for biodiversity. Regression models based on measures of habitat structure explained 74–86% of the variance in macrobenthic diversity, suggesting that removal of habitat structure in relatively low-structure soft-sediment systems can significantly decrease their biodiversity, and consequently that of the wider marine ecosystem.

Morphological traits that affect ecological performance can often be employed as surrogates for niche attributes (but see Section 4.3.4). The potential effect of habitat complexity on the structure and the composition of ecological communities is often correlated with the surface area of that habitat, but very few studies have actually succeeded in disentangling both components (Baldi 2008). But whether habitat complexity or the interaction between habitat complexity and the surface area of a habitat is the main driver in structuring community, both theories predict an increase in species and community diversity.

Habitat complexity can be geological or biological. In addition to the topographic features of the seafloor (e.g. sand waves, rocky outcrops etc.), many sessile, habitat-forming species (e.g. *Lophelia*, *Pentapora*, *Modiolus*, *Sabellaria*, macrophytes etc.) or bioturbating species (e.g. *Nephrops*, *Goneplax*) are important ecosystem/habitat engineers (also: bioengineers) that can form discrete and complex habitats that provide a range of microhabitats and increase local biodiversity. The diversity of some of these complex biogenic habitats can be more problematic to quantify in the field (see Sections 2.5 and 2.6).

3.5 Resilience

The resilience of an ecosystem describes the capacity of a system to absorb disturbance and reorganize to retain essentially the same function, structure, identity, and feedbacks (Walker & Meyers 2004). The likelihood of regime shifts may increase when resilience is diminished either by reduced diversity or by removal of functional groups of species or trophic levels (Folke *et al.* 2004 and references therein).

With this respect, functional diversity describing the physiological capacity of a community complementing phylogenetic diversity is an important component of biodiversity. Phylogenetic diversity may also influence some of the functional diversity measures (Petchey & Gaston 2002). Until recently, the number of functional groups represented by species in a community can be used to calculate functional diversity on the basis of traits (Schratzberger *et al.* 2007). Over the last decade, however, several functional diversity metrics have been developed. Unfortunately, it is still unclear if they are properly measuring what they are supposed to measure (Bremner *et al.* 2003). These techniques have been restricted to certain parts of biodiversity so far. However, novel techniques are developed enabling the analysis of functional diversity in microbial communities (Schloss & Handelsmann 2008).

It has often been suggested that a generally higher biological diversity increases the resilience of an ecosystem. However, this is still discussed among ecologists in the so called diversity-stability debate (McCann 2000).

Nevertheless there is strong evidence that redundancy in biodiversity favors resilience of ecosystems and that environmental conditions influence functional complementarity in structuring communities (Hooper *et al.* 2005). This is supported by findings illustrating that decreased phylogenetic diversity of microbial communities result in a decreased number of functional niches (Torsvik *et al.* 1998). Furthermore, there may be phylogenetic groups that can be lost from some ecosystems without substantial alterations of ecosystem function, as two species can show similarities in the way they metabolize, reproduce etc. They probably do not carry out these activities in exactly the same way or at the same time, and the functional significance of these differences is likely to depend on the species and ecosystem in question (Snelgrove 1999). Response diversity is based on differences in the capability to respond to environmental stresses and disturbance within phylogenetic groups analogous in their ecological functions thereby conferring resilience on the community (Naeem & Li 1997).

3.6 Rarity

Rarity is an important criterion in assessing biodiversity. Rare species may be prone to human impacts and eventually to extinction and are therefore of special concern in the management of marine ecosystems. The criterion 'rarity' has been applied in the listings of threatened species for various marine regions, e.g. by OSPAR and HELCOM.

However, as emphasised by the Working Group on Fish Ecology (ICES, 2003), rarity can also be a natural phenomenon, and many species of an ecosystem can be considered 'rare'. Natural rarity has two major components: numerical abundance and spatial extent of distribution and should be evaluated at the stock or population level if there are concerns with regards genetic diversity.

Rarity can be due to (or exacerbated by) anthropogenic causes, but it can also be a misperception, and distinguishing between the basis for our perception of the 'rarity' of a species is important in relation to advising on biodiversity. Some species may appear rare, but this may be related to inadequate sampling of the species. Our perception of rarity is strongly affected by sampling protocols (e.g. time, gear, area of survey). If the appropriate sampling method is used on the appropriate spatial and temporal scale, then a species may no longer be perceived as rare.

Rabinowitz (1981) described (for plants) seven categories of rarity based on species abundance, geographical distribution and habitat specificity. These categories were:

- Wide range, broad habitat specificity, local population size somewhere large.
- Wide range, broad habitat specificity, local population size small everywhere.
- Wide range, restricted habitat specificity, large populations in habitat.
- Small range, broad habitat specificity, populations somewhere large.
- Small range, restricted habitats, populations somewhere large.
- Small range, broad habitat, small populations.
- Small range, restricted habitats, small populations

The combination of abundance and distribution as important elements of rarity is also highlighted by OSPAR, who defined "A species is rare if the total population size is small. In case of a species that is sessile or of restricted mobility at any time of its life cycle, a species is rare if it occurs in a limited number of locations in the OSPAR Area, and in relatively low numbers. In case of a highly mobile species, the total population size will determine rarity".

Many of the taxa recorded in field surveys may only be recorded occasionally, whether this is at a few sites and/or in low numbers, and so may be perceived as 'rare'. Such 'rarity' should be viewed in the context of the survey (e.g. gear type, catch efficiency, survey grid, sampled area) so as to better inform on whether a species is rare or just ineffectively sampled, as an artefact of the survey design.

For example, Figure 3.1 shows the frequency of occurrence of fish and shellfish species and the total number of each species caught in a single survey. In general there is a relationship between these values, with the most abundant species also the most frequently occurring. However, a few species, such as pilchard and anchovy, can be found in relatively large numbers but across comparatively few sites.

In terms of data analysis, there may be some debate with regards to 'filtering' of data. The inclusion or exclusion of 'rare' species from field data for deriving biodiversity metrics should be both documented and justified. Whilst it may be justifiable to exclude those species for which the survey is known to be ineffective, excluding species on the basis of a low occurrence and/or abundance may be counterintuitive to examining 'biodiversity'.

The implications of the inclusion/exclusion of 'rare' species in biodiversity studies needs to be better examined, as there may be a trade-off between amount of biodiversity information and accuracy/precision of any resultant metric, and this should be examined with proper statistical tools (e.g. see Cunningham & Lindenmayer 2005; Ellingsen *et al.* 2007; Cucherousset *et al.* 2008; Fontana *et al.* 2008).

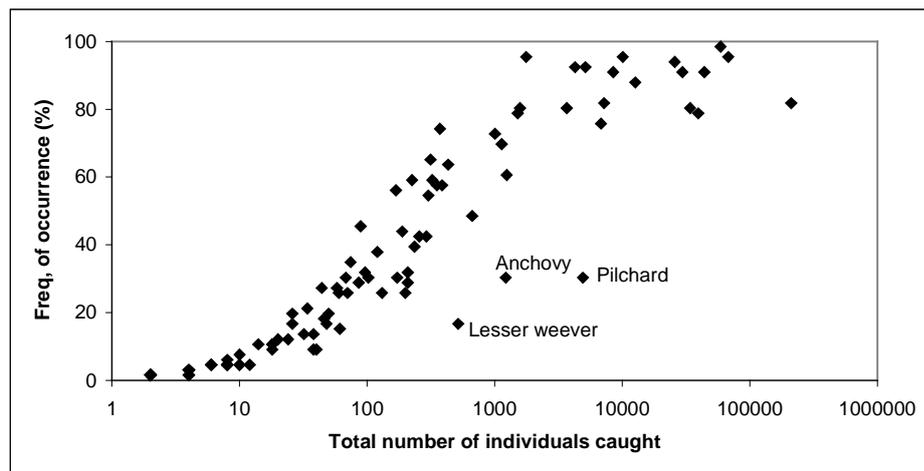


Figure 3.1. Frequency of occurrence and total number individuals caught for 104 fish and shellfish species taken in the UK Q4 IBTS survey in the Irish and Celtic Seas in 2009, highlighting the dominance and ubiquity of comparatively few species.

3.7 Extinction and extirpation

The term extinction is often used in varying meanings. For the purposes of the present report, we use the term extinct to identify those species that have died out over their entire, global range. Although some workers refer to 'local extinctions', we use the term extirpation to refer to the disappearance (due to anthropogenic activities) of the population of a species from a defined area, region, or habitat.

Extinctions of marine and diadromous species have been documented, although the exact number of extinctions is debatable (see Carlton *et al.* 1999; Dulvy *et al.* 2003 and Monte-Luna *et al.* 2007 for discussion). Among the more obvious worldwide extinctions are some marine mammals, such as Steller's sea cow and Caribbean monk seal, and seabirds such as the great auk, which were caused by human exploitation. However, extinctions among other marine taxa may be undocumented and an imperfect knowledge about marine biodiversity may lead to an underestimation of the phenomenon (Powles *et al.* 2000). Some recent studies suggest that widespread extinctions in the marine realm could soon occur (Malakoff 1997).

Although there have been relatively few documented extinctions, there are many instances of local or regional extirpations in the scientific literature. Extirpations are probably one of the main causes of biodiversity loss, both through the loss of regional species diversity and overall loss of genetic diversity. For example, Wolff (2000) reported that 10 species of algae, 10 invertebrates, 13 fishes, 5 birds and 4 marine mammals were likely extirpated in the Dutch Wadden Sea, as a result of habitat destruction and overexploitation. It should be noted that some of these species are still present in other areas (such as the wider North Sea). This highlights that any assessment of biodiversity status and of the rate/extent of biodiversity is always dependent on the spatial scale viewed.

Some of the intrinsic factors associated with reported extinctions and extirpations include specific life-history characteristics (e.g. low fecundity, late age at maturity, low mobility), restricted distribution, low mean abundance, high habitat specificity, ecological specialization, aggregating natures and high susceptibility to harvesting (e.g. McKinney 1997; Powles *et al.* 2000; Garcia *et al.* 2008; Field *et al.* 2009).

3.7.1 When is a species extinct?

In the red list categories and criteria the IUCN (IUCN 2001) states “A taxon is Extinct when there is no reasonable doubt that the last individual has died. A taxon is presumed Extinct when exhaustive surveys in known and/or expected habitat, at appropriate times (diurnal, seasonal, annual), throughout its historic range have failed to record an individual. Surveys should be over a time frame appropriate to the taxon’s life cycle and life form.” This definition relies critically on the concept of reasonable doubt on whether to determine if a species is extinct. The question of whether to presume a species extinct then becomes largely an issue of statistics, survey effort, and of prioritisation of funds.

With regards to statistical methodology, we point to Solow (1993) and Rivadeneira *et al.* (2009) as a starting point for questions on when to infer an extinction. With regards to survey effort and prioritisation of funds, these are partly statistical, but also partly societal questions. We point to Possingham *et al.* (2001) for general guidelines on informed decision making in conservation science, and to Regan *et al.* (2006) as a starting point for cost-effective decision making in the face of uncertainty on species presence. Although this last study addressed problems of eradication of an unwanted species, the methodology developed in this study might be adapted towards cost-effective assessment of extinction/extirpation.

3.8 Endemism

Species endemism or biological uniqueness refers to species found only in a particular area or of extremely restricted geographic range with narrow habitat specificity. Problems of identifying endemic species may emerge in under-explored areas such as seamounts, where the absence of evidence for a species does not necessarily mean its absence. It is thus of prime importance to keep this restriction in mind when endemism is considered.

The richness of endemic species is often recognized as an indication of the distinctiveness of certain local faunas and genetic diversity (Dolan *et al.* 1999). Thus endemic-rich areas displaying high species richness have been recently addressed as ‘hotspots’ for biodiversity in the literature and they often figure among key criteria in the establishment of conservation priorities (Roberts *et al.* 2002). Hotspots are generally thought as threatened or vulnerable areas, as the loss of endemic species are usually given more weight than the loss of non-endemic species.

Hotspots may or may not figure among the most productive areas and there is no agreed position on whether increasing biodiversity actually increases productivity in marine environments. Much of the theoretical and experimental studies regarding the relationship between biodiversity, endemism and productivity have emerged from plant ecology. Whether these actually hold in the marine environment, is notably the scope of functional ecology studies.

3.9 Large scale patterns of diversity in the sea

ICES area covers an extensive part of the Atlantic Ocean that spans a wide range of both depths and latitudes. Thus the existence and causal processes responsible for large scale patterns of diversity are of interest to ICES-based biodiversity research. Two basic large scale patterns have been recognized for the marine biota: 1) latitudinal patterns, and 2) bathymetric patterns.

The latitudinal cline of the diversity (i.e. gradual decrease of number of species from tropics poleward) is a well recognized pattern in terrestrial ecology. A similar pattern

has been postulated for marine biota. Sanders (1968) observed a difference in diversity of macrobenthos between tropical and boreal shallow seas and explained it by his 'time-stability theory' – he perceived tropics as 'biologically accommodated' systems that experienced stable conditions for a long time, while boreal communities as 'physically controlled' by environmental stressors. Several other causal mechanisms have been proposed to explain the latitudinal cline of diversity of the sea including (a) Rapport's rule (latitudinal differences in species range sizes - smaller geographic ranges in the tropics permitting more species to co-occur in a given area; (b) species-area hypothesis – based on the species-area relationship; (c) differences in geological history (relatively young geological age of polar environments); (d) gradients in solar energy input/levels of productivity; (e) 'mid-domain effect' - which assumes denser species packing around the geographic midpoint of distribution; and (f) differences in rates of evolutionary processes (speciation and extinction) (Roy *et al.* 1998, Gray 2001, Crame 2000).

While the hypotheses for potentially causing the latitudinal cline are being discussed, the generality of the pattern across different marine biota and habitats has been questioned by a number of recent studies. Clear gradients of increased diversity from the North Pole to the tropics were recorded for prosobranch molluscs (along the Pacific and Atlantic coasts of North America, Roy *et al.* 1998), bivalves (on North Pacific continental shelf, Jablonski *et al.* 2000), bryzoans (Clarke and Lidgard 2000), and various deep-sea taxa, including foraminiferans, cumaceans, isopods, gastropods and bivalves (Culver & Buzas 2000; Rex *et al.* 2000; Gage *et al.* 2004). These patterns contrast with no latitudinal effect recorded in infaunal materials encompassing Norwegian continental shelf (Ellingsen and Gray 2002) or the large part of European continental shelf waters covered by MarBEF database (Renaud *et al.* 2009).

Although it can be hypothesised that biodiversity is greater at lower latitudes, it should be recognised that this rule does not seem to apply in the North Sea (ICES subarea IV), as several Lusitanian marine species may enter the North Sea via the English Channel as well as the northern North Sea.

Another large scale diversity pattern was attributed to depth gradient. Sanders (1968) 'time and stability hypothesis' predicted that the diversity will increase with depth as deep ocean habitats provide more stable environmental conditions. The parabolic pattern (with highest diversities at intermediate depths of 2000–3000 m) was documented for polychaetes, gastropods, protobranchs, cumaceans, invertebrate megafauna and fish megafauna in the northwestern Atlantic (Rex 1983) and seemed to be a predominating depth related pattern in the seas. That pattern was related to bathymetric changes in predation, disturbance level and productivity (Rex *et al.* 1997). The increase of macrobenthic diversity towards the intermediate (2000–3000 m) depths was not observed, for example, in the Greenland Sea where the ecological patterns were masked by the historical and geographical constraints (Włodarska-Kowalczyk *et al.* 2004).

Several other ecological rules may apply to the marine fauna in the ICES area, and there is the potential for such rules to affect diversity metrics or, at the very least, should be considered when looking at metrics from different ICES ecoregions or bathymetric environments.

Polar gigantism (i.e. species in northern latitudes can attain larger body sizes) has been documented for several taxa, and Bergman's Rule suggests that the size of a species increases with latitude (and cooler environmental conditions). Hence, for any length or size-based community metrics (or analyses utilising L_{max}) it should be noted

that a single L_{max} may not apply over the entire ICES area for some species or between surveys.

In terms of shelf-dwelling fish species, juveniles tend to occur in shallow inshore waters, with fish moving offshore into deeper waters as they attain a larger size ('Heinke's law'). Such patterns are known to occur in various flatfish and skate species, and so may have implications for potential diversity metrics.

Abundance-range size relationships are documented for various fish species, both on an inter-specific and intra-specific basis. For example, species having an increased abundance tend to be more widespread. Conversely, as a population declines, the distribution can contract. This has been well documented for various fish species (see Section 3.2 of ICES, 2004; Section 7 of ICES, 2005; Section 3 of ICES, 2006).

3.10 References

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4 Biodiversity indices and indicators

4.1 Definitions and Background

Criteria

A criterion is here considered to be a measurable single characteristic (e.g. abundance) of any component of biological diversity (e.g. of a species or habitat).

Indices and metrics

For the purpose of this report these terms are used interchangeably, although there can be subtle differences in definitions, which may depend on discipline, and whether multiple criteria are involved.

An index/metric is a more or less complex measure that is often derived from several different criteria. In ecology, they are frequently used to inform on the amount of variety in a given area/time. In terms of biodiversity, the degree of variety can be assessed on various levels, e.g. at the level of species, genes or habitats. Most commonly, biodiversity is measured on the level of species.

Indicator

An indicator is one of the above measures that closely responds (in space and time) to a particular pressure and so allows managers to discriminate between natural factors and the anthropogenic pressure assessed.

The assessment of environmental state provided by one or more indicators should allow inferences to be made on the wider state of biodiversity components in that ecosystem. State means the actual (measured or otherwise assessed) environmental condition (e.g. of a species, species group, assemblage, community or habitat) in a given geographical area. The assessment of state can be derived by direct measurements of the particular biodiversity component (state indicators) or indirectly by measuring the prevailing pressures (pressure indicators).

Indicators have been widely evaluated by various ICES Expert Groups, and there are several criteria that need to be considered when trying to determine the utility and applicability of an indicator (Table 4.1).

4.2 Previous and current applications of marine biodiversity indicators

Although the 2010 CBD target has not been widely evaluated in the scientific literature (e.g. Fontaine *et al.*, 2007), various bodies have discussed or illustrated various types of biodiversity indicators or marine ecosystem indicators (e.g. Rogers & Greenaway, 2005; Defra, 2009).

4.2.1 United Nations Environment Programme (UNEP)

UNEP produced some guidance (Bubb *et al.*, 2005) on the types of biodiversity indicators that could be used, which for marine taxa could include:

- Change in extent of selected biomes, ecosystems & habitats
- Change in species abundance and distribution
- Coverage of protected areas
- Change in the status of threatened species
- Marine Trophic Index

- Nitrogen deposition; numbers and costs of alien invasions
- Connectivity and fragmentation of ecosystems
- Health and well-being of people in biodiversity-dependent communities

4.2.2 SEBI 2010

A Pan European initiative, SEBI 2010 (Streamlining European 2010 Biodiversity Indicators), was launched in 2004. Its aim is to develop a European set of biodiversity indicators to assess and inform about progress towards the European 2010 targets.

The work is performed in collaboration between the EEA (the European Environment Agency), DG Environment of the European Commission, ECNC (the European Centre for Nature Conservation), UNEP/PEBLDS Secretariat with the lead of Czech Republic and UNEP-WCMC (the World Conservation Monitoring Centre).

The ECNC (2005) and EEA suggested various 'headline biodiversity indicators', including:

Status and trends of the components of biological diversity

- Trends in abundance and distribution of selected species
- Change in status of threatened and/or protected species
- Trends in extent of selected biomes, ecosystems and habitats
- Trends in genetic diversity of domesticated animals, cultivated plants, and fish species of major socioeconomic importance
- Coverage of protected areas

Threats to Biodiversity

- Nitrogen deposition
- Numbers and costs of invasive alien species
- Impact of climate change on biodiversity

Ecosystem integrity and ecosystem goods and services

- Marine trophic index
- Connectivity/Fragmentation of ecosystems
- Water quality in aquatic ecosystems

Sustainable Use

- Area of forest, agricultural, fishery and aquaculture ecosystems under sustainable management

4.2.3 OSPAR EcoQOs

The OSPAR EcoQO system has been developed by OSPAR in collaboration with ICES through a pilot project in the North Sea. EcoQOs provide a means by which OSPAR Contracting Parties in the North Sea define desired qualities of the marine environment and identify the effectiveness of their management of human impacts. EcoQOs can take the form of targets, limits or indicators. The set of EcoQOs used in the pilot phase include (for full list and further details see OSPAR 2007):

- Biomass of commercial fish species
- Seal population trends
- Bycatch of harbour porpoises

- Oiled Guillemots
- Plastic particles in the stomachs of seabirds
- Imposex in dogwhelk *Nucella lapillus*
- Threatened and declining species
- Contaminants in seabird eggs and feathers
- Changes in the proportion of large fish
- Local sandeel availability to black-legged kittiwakes
- Seabird population trends

4.2.4 Marine Strategy Framework Directive (MSFD)

The Marine Strategy Framework Directive is in force since June 2008 and establishes a framework within which EU Member States shall take the necessary measures to achieve or maintain Good Environmental Status (GES) in the marine environment by the year 2020. GES is defined by 11 environmental descriptors that also represent the framework for the development of indicators. Several descriptors are directly related to the status of biodiversity (in particular D1, D4 and D6, see ICES 2009a). The EU commission will provide guidance on criteria and methodological standards by July 2010, based on recommendations given by dedicated expert groups under ICES and JRC.

Task Group 1 (Descriptors of Biodiversity) is expected to finalise recommendations this spring.

4.3 Metrics for species and species diversity

There are various methods of assessing temporal and spatial patterns in species diversity, and which may be suitable for developing indicators, and these can be categorised as (a) species-specific indicators (e.g. sentinel or surrogate taxa), (b) community metrics, (c) taxonomic diversity metrics, (d) functional diversity metrics, (e) size-based indicators, and (f) food-web indicators.

There are many ways one can estimate biodiversity based on species-related indicators. Species richness or the number of functional groups, for instance, has the advantage of being simple to understand and to calculate. However, they may give an incomplete view of species diversity. A potentially optimal strategy would be to assess and correlate both types of indicators to apprehend different aspects of community functioning (Micheli and Halpern 2005).

4.3.1 Species-specific metrics

Species-specific indicators, including sentinel species have several possible roles in monitoring programmes. Sentinel species may be those species that are sensitive to a particular anthropogenic disturbance. Hence, if such a species is present in an area, it is assumed that the current levels of disturbance are insufficient to have affected the species, at least at the population level, and it is so presumed that other elements of that community are also at an acceptable level (i.e. the sentinel species is viewed as a proxy for the overall community, and if it is present, or at a certain level of abundance, then ipso facto, the overall community is doing well). Sentinel species may be sensitive, fragile or threatened species, and data for such species (e.g. their presence, abundance or extent) may facilitate more focused sampling/monitoring programmes that are cost-effective. Their utility may, however, depend on the population status of

the species, and the statistical power to detect significant trends in very rare species may be problematic.

4.3.1.1 Vulnerable or Threatened and Declining species

Assuming that species listed as being 'threatened' (e.g., listed on IUCN red lists, in the Annexes of CITES, in the 'Threatened and Declining' lists by OSPAR or HELCOM, or on national wildlife legislation) are those that are most at risk of extirpation, then indicators that inform on the status of such species may serve as a useful warning system for the prevention of biodiversity loss. However, it should be noted that:

- The selection criteria for species to be listed as 'Threatened and declining' are debatable and not all taxa are appraised. For example, larger and more 'charismatic' megafauna may be selected more than little known cryptic species.
- Existing surveys may be insufficient to inform on the status of the species in question if sampling effort is low in the species habitat/remaining range, although some national (or regional) surveys may be appropriate.
- If a species is very rare, then even a dedicated survey may have a low precision (and may be damaging to the population).
- There needs to be due consideration of how to incorporate other sources of information and 'local knowledge'.

In both national and international marine policy, trends in threatened species are increasingly used as biodiversity indicators. For example, HELCOM has developed a respective indicator under the Baltic Sea Action Plan and IUCN led the development of the Red List Index (RLI). The RLI is used to illustrate the overall rate at which sets of species change in overall threat status (i.e. projected relative extinction risk), based on population and range size and trends as quantified in the categories on the IUCN Red List.

4.3.2 Community metrics

Community metrics (e.g. species diversity, species richness and evenness of the community) are the traditional metrics of biodiversity that have been used in many studies. Such indices are gear-specific, are heavily dependent on sample sizes (e.g. recruitment events) and will vary naturally on both temporal and spatial scales (e.g. with sediment and depth). Community-based metrics may be highly variable and so reducing a species-abundance matrix to a univariate index of diversity may not identify a significant change, even if there has been a change in species composition. Although providing useful spatial indices for those faunal groups in the community sampled by a particular gear, it can be difficult to compare results with those from other studies that have used different gears or protocols.

On a more practical note, taxonomic problems may exist for certain groups of organism, and the resource required to ensure accurate identification may need to be considered. Although it is in many ways admirable to attempt to determine the total species diversity, many studies have reported that the effects of disturbance can be apparent with sub-sets of species, which may be the more frequently occurring (or better known) taxa or simply higher taxonomic groups (see Section 4.3.7 for further discussion).

4.3.3 Taxonomic diversity metrics

Traditional indices of species diversity provide no information on the phylogenetic diversity of samples. It is, however, well documented that disturbed systems may have a large number of closely related species, and less disturbed systems have species representing a broader spectrum of higher taxa (e.g. Warwick & Clarke, 1995; Somerfield *et al.*, 1997). Indices of taxonomic distinctness and taxonomic diversity have been developed to account for this.

Taxonomic diversity (Δ) is a univariate metric that gives the average “taxonomic distance” between any two organisms in the sample (e.g. the length of the path based on the phylogenetic tree). This metric incorporates elements of phylogenetic relatedness and the evenness of the sample. Taxonomic distinctness (Δ^*) is a univariate index that calculates the average “taxonomic distance” between species in a sample (Clarke and Warwick, 1999), therefore providing a purer index of taxonomic relatedness (i.e. phylogenetic diversity). The merits of this index are that it may also provide an indirect index of functional or trophic diversity, it is not heavily influenced by sampling intensity (viz. community metrics), may be insensitive to habitat issues, and can be used with presence/absence data (Somerfield *et al.*, 1997; Clarke and Warwick, 1998; 1999; Warwick and Clarke, 1998).

One of the potential limitations of the method is that taxonomic classification schemes, which are in many cases arbitrary, may not be consistent between higher taxa, and so the application of this method to the entire faunal suite of organisms in a sample may introduce some taxonomic artefacts (Clarke and Warwick, 1999). Nevertheless, for those taxa for which the taxonomic nomenclature is well described and accepted by the scientific community, indices of taxonomic diversity may provide a useful method of observing temporal-spatial changes in communities (e.g. Hall and Greenstreet, 1998; Price *et al.*, 1999; Rogers *et al.*, 1999; Mistri *et al.*, 2000). Hall and Greenstreet (1998) applied these methods to fishes in the northern North Sea, and observed that changes in taxonomic distinctness and taxonomic diversity mirrored changes in traditional community indices. Clarke and Warwick (2001) subsequently provided an additional complementary index called the “variation in taxonomic distinctness” (VarTD, Δ^+). For example, a sample of species representing different orders of organisms, would give a higher Δ^+ than a sample of species representing different families or genera within a single order.

The utility of taxonomic distinctness indices in long-term monitoring studies has been tested by Warwick *et al.* (2002) who applied these methods to a benthic monitoring survey of Tees Bay, UK. This study showed that it can be very useful in situations when traditional diversity metrics can give a non-significant or misleading signal. Samples included in this analysis were collected from 1973–1996, and this period encompassed the mid to late 1980s - the time of an ecosystem shift in the North Sea. Different effects of the shift were recorded for the Tees Bay benthos when both traditional and taxonomic distinctness indices were applied to monitoring data. The values of Shannon-Wiener index increased (resulting from the decreased abundance of some dominant species) giving a false impression of improving environmental quality, while the significant decline in average taxonomic distinctness values showed that the taxonomic spread of the benthic fauna largely decreased after the regime shift (Warwick *et al.* 2002).

The major advantage of the taxonomic distinctness approach is that it allows scientists to monitor the status of the environmental quality without setting a reference ‘pristine’ site (Leonard *et al.* 2006). The observed values of the taxonomic distinctness

indices can be compared to the expected values that (together with upper and lower 0.95 probability limits) are derived from randomization of the subsets of the regional species lists. Thus instead of relating the observed situation to a spatial 'reference site' it can be compared to a 'reference condition'. Leonard *et al.* (2006) analyzed the average taxonomic distinctness of groundfish data collected within the ICES quarter rectangles west of British Islands and found that the points that fell below the 95% probability limit of expected values represented areas of higher beam-trawling effort.

4.3.4 Functional diversity metrics

The major drawback in classical and taxonomic-based indicators is to consider that all the species within a community are equal and thus functionally interchangeable. In nature and specifically in marine ecosystems, this is of course, unlikely to be the case. Another family of biodiversity indicators, the functional-based indicators, is measured by the range of species traits (e.g. morphological or ecological) in a community and therefore assumes that species identity is an important component of communities. These indicators are either based on traits referring to the ecological role played by species within a community (effect traits) and/or the responses of species to specific perturbations (response traits). Petchey and Gaston (2006) recently categorized them into two groups: discontinuous or continuous measures.

Discontinuous measures are the ones that are based on predefined groups such as trophic or ecological guilds whereas continuous measures are based on quantitative or qualitative information about functional traits of species. The similarity among species is represented in a n -dimensional trait space and derived measures either related to that space (e.g. pair-wise distances) or classifications from that space (e.g. branch length) define the continuous diversity metrics.

In discontinuous measures, the density (or extent) of particular functional (or taxonomic) groups may be a useful method of monitoring certain aspects of the function of the community (e.g. the densities of fragile/sensitive or habitat-forming species, the densities of opportunistic and scavenging species). The basis for this is that the species composition of a disturbed habitat is often dominated by opportunistic species (e.g. r -selected species that are fast-growing species with a high reproductive potential), whereas a climax community will contain a wider spectrum of taxonomic and functional groups, with more habitat-forming and k -selected species.

The decision of assigning a species into a group, i.e. determining that a species is dissimilar enough to be classified into a distinct group, remains largely arbitrary or subjective. Indices derived from discontinuous measures are however less 'effort-consuming' than the indices estimated from continuous measures as they do not imply quantitative measurements of species traits. They can therefore be estimated using (as the majority of studies do) generic information such as the trophic guilds or species vulnerability to oxygen concentrations.

Whilst there is an ever-increasing number of continuous indices of functional diversity, their explanatory power and responses to anthropogenic and natural pressures remain largely unexplored, especially with regards field data (Brind'Amour 2007). The rationale behind these biodiversity indices is that anthropogenic pressures would likely decrease the number of functional traits within a community and therefore modify and presumably reduce the measure of functional diversity (e.g. convex hull volume, Petchey and Gaston 2002; tree branch length, Villéger *et al.* 2008). Concurrently to taxonomic diversity, functional diversity indices may also include evenness,

therefore incorporating information of how abundance (or biomass) is distributed among species traits (e.g. see review by Laliberté and Legendre 2010).

Limitations of functional diversity include

- Some aspects of the life history or ecological role of many species can be poorly known
- For those taxa where morphological traits are used to inform on function, then analyses may simply reflect taxonomic differences

4.3.5 Size-based indicators

Size-based indicators or any life history traits may also refer to functional aspects of fish species. They have a relatively sound theoretical background, with impacted sites tending to have more, smaller-bodied species, and pristine sites (which may correspond to “climax communities”) having an increased proportion of large-bodied species/individuals (Rochet *et al.* 2000), and the proportion of large fish has been used as an OSPAR EcoQO for the fish “community”. Although size-based metrics are well documented for fish, there have been some studies on the size spectra for other marine taxa (e.g. Duplisea, 2000; Cavender-Bares *et al.*, 2001; Queiros *et al.*, 2006).

Some of the practical considerations for size-based metrics include:

- Are the largest species/individuals sampled effectively in existing surveys?
- How to best address recruitment pulses

Extensive analyses of size-based metrics have been undertaken for fish assemblages, and so the reader is referred to the work of the ICES Working Group on Fish Ecology for further information (ICES, 2003–2008, 2009b).

4.3.6 Food-web indicators

Trophodynamic indicators are estimated from network or food-web analyses, such as the Marine Trophic Index (Pauly & Watson, 2005; Bhathal & Pauly, 2008) and other trophic and trophodynamic indicators (Cury *et al.*, 2005; Gascuel *et al.*, 2005).

4.3.6.1 Marine Trophic Index

There has been an increased interest in the use of the Marine Trophic Index, although WGBIODIV considered there still needs to be a better evaluation whether such an index can act as an indicator. Such issues to be better addressed include:

- The MTI based on commercial landings is biased towards the (commercially exploited) fish (and shellfish) stocks and so may or may not be informative for wider ecosystem state.
- Landings may not be representative of the populations in the study area, given the selectivity of gears, fisher behaviour and market demands.
- There have been temporal changes in landings categories, and so what are the effects of the increased use of species-specific landings categories? Additionally, temporal changes in quotas and/or management plans may have an effect on such an index.
- Some species of lower trophic level (e.g. planktivorous fishes, bivalves) may be landed in variable quantities, and does this unduly affect such an index? It should be noted, however, that some studies omit species below a particular trophic level.

- The MTI based on fishery-independent groundfish surveys may not be informative for larger, higher trophic level fish, as some such species may only be caught in small numbers. Catches in surveys will also have issues regarding survey design (gear, grid etc) and some 'demersal' surveys use high headline trawls to allow sampling of (lower trophic level) pelagic species. Hence, analyses of survey data need to have appropriately filtered data.
- The trophic level of fish can vary spatially, temporally and ontogenetically, and there are some cases where the published trophic level (as predicted from the observed $\delta^{15}\text{N}$) seems contrary to their position in the overall food web. For example, some studies (e.g. Pinnegar *et al.*, 2002) have reported that spurdog *Squalus acanthias* (3.41) is at a lower trophic level than herring *Clupea harengus* (3.79) and sprat (4.09), despite being an important predator of both these clupeids. Similarly, megrim *Lepidorhombus whiffiagonis* (3.56) was assessed to be at a lower trophic level than witch *Glyptocephalus cynoglossus* (3.88), despite the former being more piscivorous. Hence, there may still be some need to better reconcile trophic level with the role in the food web.
- The MTI was designed to correspond with the perceived reduction in top predators, but 'threat' is not solely related to high trophic level. Other factors can confer susceptibility to threat (e.g. high habitat specificity, restricted distribution). Indeed, some fish species of conservation concern may have a relatively low trophic level², including shad *Alosa* spp. (3.6), Atlantic sturgeon *Acipenser sturio* (3.5), smelt *Osmerus eperlanus* (3.0) and wolf fish *Anarhichas lupus* (3.2). Similarly, in warm temperate to tropical seas, there has also been concern over some herbivorous fishes.
- It is questionable whether high trophic level is related to 'vulnerability' in other ecological or taxonomic groups. For example, some scavenging and predatory invertebrates may be less sensitive to fishing impacts than to some fragile bodied filter feeding invertebrates
- It is unclear whether trophic groups are consistently responsive to a particular pressure over the ICES area
- The reliance of looking at top predators to inform on the trophic structure, whereas primary production etc. may also be informative about the ecosystem.

4.3.7 Surrogate methods

The identification of all organisms in the collected material to a species level is labour intensive, time consuming and requires a considerable level of taxonomic expertise. Rapid pollution assessment impact programs fuelled the search for cheaper and quicker methods. Surrogates are defined as quantities that are more easily determined than species-level community patterns, but which correlate strongly to them (Olsgard & Somerfield 2000). Two basic surrogate methods have been proposed: 1) to lower the taxonomic resolution of the analyses or 2) to analyze only a selected 'surrogate' group that is treated as an indicator of the patterns of the wider group of organisms (e.g. of the whole macrobenthic community).

² Trophic level values taken from FishBase.

Gaston (2000) claimed that numbers of higher taxonomic groups can be good estimators of species richness. The term 'taxonomic sufficiency' was firstly introduced in pollution effect studies and was defined as the lowest level of taxonomic resolution required to detect pollution-related changes in benthic communities (Ellis 1985). The obvious advantages of the taxonomic sufficiency approach are cost and time savings. Identification to family or order level can reduce the costs by 50 and 80 % (Ferraro & Cole 1995) and sample processing time by 40 and 76 % (Thompson *et al.* 2003) when compared to species level identifications. The analyses of higher level taxonomic groups can also reduce the risk of potential taxonomic classification errors as higher groups are often more clearly defined than are species, which can be especially advantageous when several datasets produced by different laboratories of inconsistent taxonomic expertise are to be merged. And finally, lower taxonomic resolution analysis can be the only feasible solution for analyzing materials containing poorly described taxonomic groups or sampled in remote and poorly described geographical regions.

Families have been shown to be a good predictor of species diversity and distribution patterns in a number of macrozoobenthos studies. These included surveys of benthic fauna in various habitats (soft bottom, kelp beds, pebble-sandy beaches or mussel beds) and patterns related both to natural and anthropogenic processes (Table 4.2). Even phyla were proved to be useful in detecting strong pollution effects.

Gaston (2000) listed a number of situations in which higher taxa are unlikely to be good surrogates of species richness. These include: 1) number of higher taxa very similar to number of species (minimal time and cost savings), 2) number of higher taxa extremely small relative to number of species and 3) studies combining data from regions largely differing in basic taxonomic structure (e.g. isolated islands with high ratios of species to higher taxa).

The second surrogate method is to select a 'surrogate group' and treat the patterns observed for this group as a proxy of the patterns of the wider set of organisms. For example polychaetes have been shown to be a good predictor of the whole soft bottom macrozoobenthic community variability (Olsgard & Somerfield 2000). That was certainly linked to the high dominance of this group in soft sediment communities, but also to the fact that they exhibit relatively high diversity of functional guilds, reproductive strategies and levels of disturbance resistance (Olsgard & Somerfield 2000). Other studies reported excellent performance of molluscs in diversity assessments of rocky shore and kelp-associated macrofauna (Smith 2005, Anderson *et al.* 2005). It seems that the predictive potential of selected taxonomic groups as surrogates for the whole macrobenthic community varies considerably across different habitat/community types.

Recently a new surrogate method has been suggested for regional biodiversity assessment. Warwick and Light (2002) proposed that death mollusc assemblages might be a useful indicator of the regional biodiversity.

The effects of certain pressures on microbial communities have been mostly studied for bacteria. Recently, it could be shown that certain pressures (e.g. contamination and physical disturbance) directly affect community structure (e.g. Bachoon *et al.* 2001; Bissett *et al.* 2007; Cappello *et al.* 2007; Choi *et al.* 2005; Edlund *et al.* 2006; McCaig *et al.* 1999; McKew *et al.* 2007; Miralles *et al.* 2007; Powell *et al.* 2005). Depending on the kind of pressure, community changes can become evident on different genetic levels. Eutrophication, for example, favours certain bacterial classes containing functional groups such as sulphate reducing bacteria (Asami *et al.* 2005; Kawahara *et*

al. 2009). However, more research is necessary to validate the diversity measures and genetic levels necessary to identify changes in either phylogenetic or functional microbial diversity.

4.3.8 Indices of Biological Integrity

There have been several instances where Indices of Biological Integrity (IBIs) or Biotic Indices (e.g. Borja *et al.*, 2000, 2003, 2004) have been proposed or established, particularly in inshore and estuarine habitats where there are clear gradients in environmental quality. Such indices may include various diversity metrics or criteria. It is unclear whether or not such approaches can be applied to more widespread, regional data sets.

4.3.9 Data issues

Irrespective of metric used, there are some fundamental considerations with regards to the analyses of station-species matrices derived from survey data, and although not addressed fully in this report, these issues need due consideration, for example:

Data filtering:

Which species (if any) should be removed? There is debate whether or not 'rare' species should be removed, but what about those species that are only vagrants to the survey area, or species that are not sampled effectively in the survey, or non-native species? Any data filtering must be documented and justified.

Data treatment:

What to do when the data matrix has individuals of 'undetermined' species as well as identified species within that family or genus?

How are recruitment pulses addressed?

How to address 'extreme' events (e.g. severe weather or oil spills) that may affect some time series of data and confound with other effects

How to treat data of varying quality, should they be excluded, or can they be used in semi-quantitative analyses?

4.4 Genetic diversity

An integral part of biodiversity is genetic diversity, which refers to the variation of genes within species. It is distinguished from genetic variability, which describes the tendency of genetic characteristics to vary. Measures of genetic diversity of a population include gene diversity, heterozygosity and alleles per locus:

- Gene diversity describes the proportion of polymorphic loci across the genome;
- Heterozygosity identifies the mean number of individuals with polymorphic loci;
- Alleles per locus demonstrate genetic variability.

The genetic structure of a population is defined by its gene pool's allele and genotype frequencies. Changes in this structure can cause microevolution by the following mechanisms:

- Genetic drift, change in allele frequencies in small populations
- Gene flow, allele exchange between two populations via migration

- Mutation, generation of new variations
- Non-random mating, change in ratio of genotypes in populations (e.g. via inbreeding)
- Natural selection, differential reproductive success

Genetic diversity and biodiversity are strongly linked in the sense that diversity within a species is necessary to maintain diversity among species, and *vice versa* (Lankau & Strauss 2007). It also plays an important role in the survival and adaptability of a species to provide potential for adaptation. Maintenance of genetic diversity is guaranteed by various mechanisms, one of which is mediated by viruses aiding in the genetic shifting process.

4.5 Habitats and biotopes

Habitat assessments may be viewed on two different levels. Firstly, the status of a defined habitat/biotope of conservation or management concern, including as a parameter the presence or absence of typical species within this habitat. Secondly, the overall diversity of different habitats/biotopes (and microhabitats) in a defined area may be of interest. WGBIODIV consider the former to be better termed 'Habitat status', with habitat diversity referring to the latter.

Habitat status may be assessed by several criteria, such as range, extent, species composition, structural features and diversity etc. Relevant criteria and assessment methods have been implemented for certain habitats covered by the EC Habitats Directive.

Further work is required to evaluate the most appropriate methods for examining habitat diversity in the marine environment. Important considerations to be addressed include:

- Habitat classification: There is a widely accepted pan-European hierarchy of habitat classification (EUNIS), although agreement on which are the most appropriate tiers within this hierarchy to be monitored and assessed needs to be agreed
- The EUNIS scheme does not include pelagic components, and so a comparable hierarchy for pelagic assemblages or communities may still be needed.
- There has been increased interest in the connectivity between important habitats (or Marine Protected Areas).
- Some structured habitats (e.g. seagrass meadows, reefs, saltmarshes) which are known to support a high diversity and have a functional importance and be susceptible to some human impacts may need to have their habitat complexity considered in any metric/index or indicator

For example, seagrass meadows are known to be complex habitats that provide a variety of microhabitats and niches, thus affecting species abundance and community richness for many taxa (Heck & Wetstone, 1977; Edgar & Robertson, 1992; Edgar *et al.*, 1994; Moranta *et al.*, 2006); in addition they can be highly effective nursery habitats providing spawning substrates, refuge from predators and trophic support for associated organisms, thereby having a potentially positive influence on biodiversity. In particular for benthic epi- and infauna, there is clear evidence for differences in abundance and diversity of small-sized moderately mobile organisms, the areas of higher shoot density exhibiting higher values in both leaf and sediment compartments than the nearby less vegetated or bare sediments (Gambi *et al.*, 1998).

Similarly, biogenic reefs formed by species such as *Sabellaria*, *Lophelia*, *Modiolus*, *Pentapora* and various sponges often have a large number of associated species (e.g. Anadon 1981; Jensen & Frederiksen 1992; Mortensen *et al.* 1995; Dias & Paula 2001; Dubois *et al.* 2002, 2006; Jonsson *et al.* 2004; Rees *et al.* 2008).

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Table 4.1 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 4.2. Taxonomic sufficiency documented for natural and anthropogenically induced benthic patterns. The taxonomic level is regarded as sufficient (+) when R is equal to or higher than 0.75 (in distribution studies: R stands for a Spearman rank correlation of relevant similarity matrices, in diversity studies, R stands for the correlation between numbers of species and the numbers of relevant taxa in the samples). The table is given after Włodarska-Kowalczyk & Kedra (2007) and the references are listed within this paper.

Type of study	Reference	Genus	Family	Order	Class	Phylum
<i>soft bottom - natural patterns</i>						
glacial sedimentation gradient	Włodarska-Kowalczyk & Kedra 2007	+	+	+	-	-
estuarine gradient	De Biasi <i>et al.</i> 2003	+	+	-	-	-
<i>soft bottom - anthropogenic disturbance</i>						
dredgings disposal	Somerfield & Clarke 1995	+	+	.	.	+
heavy metal contamination	Somerfield & Clarke 1995	+	+	.	.	+
organic pollution	Warwick 1988a	.	+	.	.	-
oil drilling pollution	Olsgard <i>et al.</i> 1997	+	+	+	-	+
sewage discharge	Thompson <i>et al.</i> 2003	+	+	+	-	.
oil drilling pollution	Olsgard & Somerfield 2000	+	+	+/-	-	-
Pulp-mill effluent	Warwick 1988b	.	+	.	.	+
sewage sludge	Warwick 1988b	.	+	.	.	+
hydrocarbon pollution	Warwick 1988b	+
heavy metal contamination	Vanderklift <i>et al.</i> 1996	+	+	+	-	.
fish farming effects	Lampadariou <i>et al.</i> 2005	+	+	+	+	+
<i>other habitats</i>						
Kelp associated fauna - domestic effluents outfall	Smith & Simpson 1993	.	+	+	+	+
pebble-sand beaches - estuarine gradient	Dethier & Schoch 2006	.	+	.	-	-
Deep-sea mussel beds - diversity assessment	Doerries & Van Dover 2003	+	+	+/-	.	.
Hard bottom – estuarine gradient	Urkiaga-Alberdi <i>et al.</i> 1999	.	+	.	.	.

5 Priority topics for WGBIODIV within the ICES Science Plan and WGBIODIV contributions for the 2010 SSGEF session and Marine Biodiversity theme session

WGBIODIV considered the potential contributions of the group to the various high priority research topics for 'understanding ecosystem functioning, as identified in the ICES Science Plan. These topics covered various issues, such as:

- Climate change processes and predictions of impacts
- Biodiversity and the health of marine ecosystems
- The role of coastal zone habitat in population dynamics of exploited species
- Fish life history information in support of EAM
- Sensitive ecosystems (deep-sea, seamounts, arctic) and data-poor species
- Integration of surveys and observational technologies into operational ecosystem
- Role of top predators (mammals, birds, and large pelagics) in marine ecosystems
- Impacts of fishing on marine ecosystems
- Carrying capacity and ecosystem interactions associated with mariculture
- Influence of development of renewable energy resources (e.g. wind, hydropower,
- Population and community level impacts of contaminants, eutrophication, and
- Introduced and invasive species, their impacts on ecosystems and interactions with
- Marine living resource management tools
- Operational modelling combining oceanography, ecosystem and population
- Marine spatial planning, effectiveness of management practices (e.g.
- Contributions to socio-economic understanding of ecosystem goods and services, and forecasting of the impact of human activities

The table provided by SSGEF was completed, and the various topics ranked as low, medium or high importance, with some topics ranked as not applicable. Some of the topics were either ambiguous or rather broad, and so were ranked as, for example, low/medium.

WGBIODIV had several discussions during the course of the meeting on potential contributions to the SSGEF session, especially with regards to 'indicators of ecosystem health' for which biodiversity issues are important elements, and for the Marine Biodiversity theme session. One abstract for a WGBIODIV-joint paper was drafted for the latter and will be finalised by correspondence and submitted.

Annex 1: List of participants

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Annex 2: Agenda

Monday 22

- Meet at 09:30
- Welcome and Introductions
- Background to WGBIODIV and ToRs
- Presentations
- Structure of report
- Draft text for ToR (a)

Tuesday 23

- Draft text for ToR (a)
- Consider the SSGEF³ Workplan, ToR (d)
- Contributions to the SSGEF session at the 2010 Annual Science Conference, ToR (e), and for the Theme Session on Marine Biodiversity
- Discussions on 'biodiversity indicators', ToR (b)

Wednesday 24

- Discussions and draft text for biodiversity indicators, ToR (b)
- Draft text for SSGEF session, ToR (e)

Thursday 25

- Finalise text for biodiversity indicators, ToR (b)
-

Friday 26

- Suggested ToRs for 2011, agree any recommendations
- Time and place for next meeting
- Close meeting (13:00)

³ SciCom Steering Group on Ecosystem Functions

Annex 3: WGBIODIV Terms of Reference for the next meeting

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

- a) Further develop unified analyses of the diversity for multiple groups (e.g. invertebrate and fish) to better examine overall biodiversity, and to compare and contrast spatial-temporal patterns in 'biodiversity' across ecological groups, with reference to ecosystem function;
- b) Further explore and assess potential biodiversity indicators, for example by undertaking comparative analyses of taxonomic, functional, surrogate and trophic metrics;
- c) Review the existing spatial approaches in assessing biodiversity status, and the spatial and temporal scales on which different elements of marine biodiversity operate, with regards the implications for survey design and indicator development;
- d) Examine the implications of survey design for estimating 'biodiversity metrics'.

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

Supporting information

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