ICES WGBIODIV REPORT 2015

SCICOM STEERING GROUP ON ECOSYSTEM PROCESSES AND DYNAMICS

ICES CM 2015/SSGEPD:04

REF. SCICOM

Report of the Working Group on Biodiversity Science (WGBIODIV)

9–13 February 2015

ICES Headquarters, Copenhagen, Denmark

ICES CIEM
International Council for the Exploration of the Sea
Conseil International pour l'Exploration de la Mer

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Recommended format for purposes of citation:

ICES. 2015. Report of the Working Group on Biodiversity Science (WGBIODIV), 9–13 February 2015, ICES Headquarters, Copenhagen, Denmark. ICES CM 2015/SSGEPD:04. 308 pp.

Contents

Executive summary

This report covers the work carried out at WGBIODIV meetings at ICES Headquarters, Copenhagen, Denmark: 18–22 February 2013, 10–14 February 2014, 9–13 February 2015, and intersessionally. Twelve scientists from eight countries, along with two members of the ICES secretariat, participated at the various meetings (see Annex 1).

Terms of Reference

No.	Term of Reference
1	For the suite of indicators and targets proposed for Descriptor 1 ("Biological diversity is maintained) by Member States in support of the Marine Strategy Framework Directive, WGBIODIV will: Consider the consequences of recent changes in environmental conditions associated with climate change on the relevance of targets set on the basis of historical data baselines. Consider the potential consequences of the introduction of management measures intended to achieve Good Environmental Status targets for particular ecosystem components on the capacity to achieve indicators/GES targets set for other ecosystem components. Assess the extent to which there are gaps in indicators and target coverage for each of the six major ecosystem components suggested by OSPAR. Evaluate how metrics and indicators for various facets of marine biodiversity can be best integrated to derive more regional and holistic assessments of 'biodiversity status'.
2	**OSPAR request 2013-3:** Support to the technical specification and application of OSPAR common indicators under D1, 2, 4, and 6. ICES will be requested to undertake an independent peer review of the technical specifications and proposed operational implementation of the indicators that will be presented. The review should consider, from the perspective of producing a set of common indicators for the OSPAR Region: whether the indicators put for-wards are appropriate to implement at a regional scale; whether the set of indicators is sufficient as a set to understand GES; identify any gaps; identify where there are difficulties in the operationalization of the indicators, with proposals for how to overcome these. Based on the outcomes of OSPAR request 2013-4 (below) (regarding maximising efficiencies for monitoring of biodiversity), identify where there are opportunities to cluster indicators that can benefit from shared monitoring/ data collection.
3	**OSPAR request 2013-4:** Provide advice on maximising the use of available sources of data for monitoring of biodiversity. Advice would be sought as to: the quality of these potential data sources and how they could be used, including but not limited to the relevance of outcomes identified in chapter 8 of the ICES MSFD D3+ report to Descriptors 1, 4 and 6.
4	**Request from SIBAS:** Identify, define and test activity-pressure-state links of indicators now and in future, including considering single/cumulative/synergistic effects of pressures.

ToRs 2 and 3 were the direct result of a request for advice from OSPAR, which required an urgent response. At the first meeting therefore, these ToRs were accorded priority. Attention was directed primarily towards ToR 2 parts (a) to (d). WGBIODIV did not feel that they were the most appropriate group to deal with ToR 2 (e), which they felt was better addressed by WGs such as WGISUR. In 2013 WGBIODIV did not feel that they had sufficient time, or the appropriate scientific expertise, to address ToR 3, and again felt that this was better addressed by WGISUR. ToR 2 parts (a) and (d) are addressed in section 2 of the report. ToR 1 part (c) and ToR 2 parts (b) and (c) are addressed in section 3. ToR 1 part (a) is addressed in section 4. And finally, Tor 1 parts (b) and (d), and ToR 4, are addressed in section 5.

ToR 2 parts (a) and (d)

The European Union (EU) Marine Strategy Framework Directive (MSFD) requires Member States (MSs) to co-operate, through the auspices of the Regional Seas Conventions, to achieve good environmental status (GES) at the regional seas scale. Where possible, MSs and non-EU countries bordering shared regional seas are requested to harmonise their management strategies and use the same indicators to monitor change in environmental status. This places considerable emphasis on having robust procedures for selecting the most effective metrics for each MSFD indicator. In this section an indicator evaluation process is developed to select a suite of effective indicators. First, twenty-three published studies considering the properties required of effective state indicators were reviewed to derive a set of 16 indicator evaluation criteria. Next, a quantitative assessment procedure was applied to these criteria to produce an overall score for each indicator evaluated. Finally, benchmark scores, by which "effective" indicators can be identified, were determined using randomised assessment simulations. The assessment procedure was trialled by assessing 33 metrics proposed by OSPAR as potential "Common Indicators" to be used by all MSs bordering the Greater North Sea. Of the 33 metrics, only 8 met the combined benchmarks for overall effectiveness, adequate spatial coverage, and being close to operational. Although WGBIODIV specifically addressed the question of selecting appropriate indicators to implement the MSFD at the regional seas scale, the indicator evaluation process is generic and can be adapted to meet other circumstances.

ToR 1 part (c) and ToR 2 parts (b) and (c)

Two types of gap in indicator coverage were examined

i) Firstly to determine whether all indicator functions provided in the EC Decisions document were covered by at least one indicator in the list of "Common Indicators" being considered by OSPAR, and if not identify which MSFD indicator functions were not being assessed.

ii) Secondly to determine whether important components or attributes of marine ecosystems were being ignored in either the list of MSFD indicator types provided in the Decision document, or the set of ecosystem components for which "Common Indicators" were being developed by OSPAR.

It was quite clear that a substantial number of indicator roles suggested in the EC Decision document (EC, 2010) were not covered by the list of proposed OSPAR "Common Indicators" (OSPAR, 2013). However, taking issues of indicator redundancy into consideration, it was also apparent that there was little need for many of these gaps to be filled. A large number of potential ecological gaps were also identified. However, it remains to be seen whether the management needed to achieve the targets set for "Common Indicators" covering the major ecosystem components might also be sufficient to address the majority of issues faced by these less represented ecosystem components.

ToR 1 part (a)

The European Union's (EU) Marine Strategy Framework Directive (MSFD) requires 'good environmental status' (GES) by 2020. Implementing the MSFD involves the use of a large number of indicators to monitor change in various attributes of a variety of different ecosystem components. For each indicator, targets representing the desired indicator value must be set: the value of the indicator expected when the attribute of the ecosystem component in question would be deemed to be at GES. However, the MSFD is cognizant of the fact that the marine environment is not a static entity, and that variation in environmental conditions can affect the state of these ecosystem component attributes. GES for each ecosystem component attribute could therefore be a moving target reflecting changes in prevailing environmental conditions.

WGBIODIV reviewed long-term changes in the marine environment of the Northeast Atlantic and presented possible future trends in environmental variables and a number of different climate change scenarios. Considerable variation in both water temperature and salinity has occurred across the region during the course of the 20th century and into the early 21st century with clear periods of lower or higher than average temperature apparent, generally associated with either reduced or enhanced salinity. These phase changes have been linked to distinct periodic events, such as the "Great Salinity Anomaly of the 1970s, or related to cyclical changes such as the "North Atlantic Oscillation" or the "Atlantic Multi-decadal Oscillation". Recent warming of waters of the Northeast Atlantic region has been associated with global climate change linked to anthropogenic use of fossil fuels leading to build up in carbon dioxide and other 'greenhouse' gasses in the earth's atmosphere. Current climate change scenarios suggest that the world's oceans, including the Northeast Atlantic, are likely to continue along a warming trend at least for the next thirty years or more.

These changes in environmental conditions have had marked effects throughout the marine ecosystems of the Northeast Atlantic. Changes related to variation in temperature and salinity among phytoplankton, zooplankton, benthic invertebrates, fish, seabird and marine mammal communities have all been observed and these are reviewed. Based on the historical changes observed, projected effects of a continuing increase in sea temperature are considered and the implications for current targets based on historical or current reference conditions for particular plankton, benthic invertebrate, fish, seabird and marine mammal indicators are assessed. Several targets may not adequately account for recent and future changes in the marine environment associated with ongoing climate change. Consequently, there is a real risk that the indicators involved may fail to meet these targets. Such failures could give the impression that management has been inadequate to achieve the goals of the MSFD. However, the alternative interpretation, that targets set for the failing indicators were inappropriate because they failed to take changes in the marine environment into account adequately and therefore did not reflect actual environmental conditions prevailing at the time of the assessment, should first be considered and discounted before introducing even more stringent management action.

ToR 1 parts (b) and (d), and ToR 4

The European Union (EU) Marine Strategy Framework Directive (MSFD) requires an "ecosystem-based approach to the management" of marine natural resources be implemented across European waters, but does not define what is meant by this term. WGBIODIV reviewed the scientific and policy-related literature addressing the development and implementation of an "ecosystem-based approach" to identify a precise terminology. Development of an ecosystem approach to management (EAM) has been an evolutionary process. The term EAM encompasses this whole process, which consists of four distinct phases, giving rise to the terminology adopted by WGBIODIV:

i) *Classical Fisheries Management* (CFM) which considers separate fish stocks in isolation of each other using simple population dynamics models (e.g. single species virtual population analysis (VPA)) utilising basic parameters for each individual stock. Only the aspirations of a single sector (fisheries) are considered.

ii) An *Ecosystem Approach to Fisheries* (EAF) takes more account of broader ecosystem processes (e.g factors affecting recruitment) and of interactions between the different targeted stocks (e.g. food web processes) that affect each stocks' dynamics. More complex models (e.g. multi-species VPA) are used, which can support management strategy evaluation (MSE), and so improve the scientific basis underpinning fisheries management decisions. Still only the aspirations of a single sector (fisheries) are considered.

iii) *Ecosystem-Based Fisheries Management* (EBFM) still only addresses the aspiration of the single (fisheries) sector, but more complex models, and relationships incorporating variables associated with other ecosystem components, are employed to estimate the impact of fishing activity on the broader marine ecosystem. This widens the perspective of MSE, allowing fisheries aspirations to be balanced against wider ecosystem consequences.

iv) *Ecosystem-Based Management* (EBM) takes EBFM to the next level by including the aspirations of multiple sectors (e.g. fishing, gravel extraction, shipping, renewable energy, etc.). At this level, MSE can address the needs of multiple sectors and impacts of each sector's activity on the marine ecosystem to maximise and balance the exploitation of marine natural resources and ensure sustainable use.

The MSFD addresses the needs of all sectors exploiting the full range of marine natural resources, but in such a way that the ecological consequences are not excessive or irreversible. The intention being that exploitation of all resources should be maintained at or just below levels that can be sustained over the long-term. The MSFD therefore requires EBM.

With each incremental phase in the development of the EAM, the level of integration required increases, but the precise shape of the integration required has been the subject of considerable debate. The literature contributing to this debate is reviewed. The conclusion that emerges is that for EBM to be made operational, a formal mechanism for processing the required integration is necessary. Just as individual stock assessments have traditionally provided the principle scientific basis supporting CFM, now a formal approach to integrated ecosystem assessments (IEA) is required to provide the scientific basis to support the EBM needed by the MSFD.

The need for, and development of, IEA has also been the focus of considerable scientific endeavour in recent years; again this literature is reviewed. A six step framework for IEA has emerged. Progress in implementing the MSFD to date is related to this framework to identify which aspects of an IEA have been completed, and to clearly establish what still needs to be done. This suggests that we are at a relatively advanced stage at scoping the IEA necessary to implement the MSFD and that good progress has been made with regard to the development of indicators and their targets, setting out the assessment process with respect to each indicator and with ensuring that the necessary monitoring programmes are in place to provide the data required to derive the indicators and carry out an assessment. However, to date little in the way of formal risk assessment has been carried out and the format of any formal management strategy evaluation is still largely undefined. The process for integrating the outcomes of individual indicator assessments has still to be decided.

Several different methods for integrating/aggregating the information conveyed from multiple assessments of a number of different individual indicators have been proposed. WGBIODIV examined some of the pros and cons associated with each method. A full IEA to meet MSFD requirements involves integration of individual assessment outcomes at several different levels:

i) from an assessment of the status of an ecosystem component based on individual indicators;

ii) to integrating these indicator assessments to derive an assessment of the status of an ecosystem component at the Criterion level;

iii) then to integrating these Criterion level assessments to produce an assessment of the status of an ecosystem component at the Descriptor level;

iv) then to integrating the these Descriptor level assessments of the status of each ecosystem component across all ecosystem components to derive an overall assessment of state at the Descriptor level,

v) and finally, potentially having to integrate these Descriptor level assessments to determine overall status of the marine ecosystem.

Two simulation exercises confirm the fact that choice of integration method affects final IEA outcomes. Deciding which integration method is most appropriate for each situation, at each level of integration, is critically important and has the potential to influence the overall outcome profoundly. It is essential therefore that these decisions are taken *a priori* of actually carrying out the assessment.

Two further areas of difficulty when undertaking IEA to support the MSFD were identified. Firstly, selection of indicators to support EBM has to date primarily been done on a case by case basis, often using a set of selection criteria to inform selection decisions. The selection criteria invariably include the need for established pressure-state relationships, so that observed changes in state can be interpreted in such a way that specific advice as to how to manage pressure can be formulated. However, these pressure-state relationships have in the most part been considered in isolation. EBM addresses the need to manage multiple activities, leading to multiple pressures, so as to achieve preconceived goals for state. Thus IEA will need to address the cumulative impacts of multiple pressures on both single state indicators and multiple state indicators in such a way that MSE can subsequently identify the appropriate 'activity mix' that gives the best overall compromise between exploitation of all marine natural resources commensurate with acceptable, sustainable levels of deterioration in ecosystem status.

Secondly, just as the pressure-state relationships have generally been considered only on an indicator by indicators basis, the approach to target setting has been similar. Targets have been set for each indicator using a variety of different baselines and reference points, with different logic often underpinning each approach. To date little consideration has been given to assessing whether the targets set for seabird indicators, for example, are compatible with targets being set for fisheries management, or for fish communities. WGBIODIV examined the targets being set for different indicators for different ecosystem components and identified several potential inconsistencies. The key 'take-home' message from this analysis is that should an indicator fail to meet its target, one should first question whether the target is actually appropriate, before immediately assuming that the management measures put in place are inadequate and placing further restrictions on the human activities involved.

1 Implementing ecosystem based marine management at a regional seas scale: identifying effective "state" indicators

In this chapter, the following Terms of Reference are addressed:

2. **OSPAR request 2013-3:** Support to the technical specification and application of OSPAR common indicators under D1, 2, 4, and 6. ICES will be requested to undertake an independent peer review of the technical specifications and proposed operational implementation of the indicators that will be presented. The review should consider, from the perspective of producing a set of common indicators for the OSPAR Region:
    a. whether the indicators put for-wards are appropriate to implement at a regional scale;
    d. identify where there are difficulties in the operationalization of the indicators, with proposals for how to overcome these.

Summary

The European Union (EU) Marine Strategy Framework Directive (MSFD) requires Member States (MSs) to co-operate, through the auspices of the Regional Seas Conventions, to achieve good environmental status (GES) at the regional seas scale. Where possible, MSs and non-EU countries bordering shared regional seas are requested to harmonise their management strategies and use the same indicators to monitor change in environmental status. This places considerable emphasis on having robust procedures for selecting the most effective metrics for each MSFD indicator. In this section an indicator evaluation process is developed to select a suite of effective indicators. First, twenty-three published studies considering the properties required of effective state indicators were reviewed to derive a set of 16 indicator evaluation criteria. Next, a quantitative assessment procedure was applied to these criteria to produce an overall score for each indicator evaluated. Finally, benchmark scores, by which "effective" indicators can be identified, were determined using randomised assessment simulations. The assessment procedure was trialled by assessing 33 metrics proposed by OSPAR as potential "Common Indicators" to be used by all MSs bordering the Greater North Sea. Of the 33 metrics, only 8 met the combined benchmarks for overall effectiveness, adequate spatial coverage, and being close to operational. Although this section specifically addresses the question of selecting appropriate indicators to implement the MSFD at the regional seas scale, the indicator evaluation process is generic and can be adapted to meet other circumstances.

Introduction

Marine ecosystems are subject to a range of human pressures, which have increased in both variety and extent over recent decades (Halpern *et al.*, 2008; Greenstreet *et al.*, 2009; Foden *et al.*, 2010), giving rise to increasing concern over the impact of this on the state of marine ecosystems (Worm *et al.*, 2006). To ensure that marine ecosystems continue to provide the goods and services on which human populations depend (Balmford *et al.*, 2002), the need for holistic ecosystem based management (EBM) of marine natural resources has become increasingly apparent (Gislason *et al.*, 2000; Sainsbury and Sumaila, 2001; Hall and Mainprize, 2004; Jennings, 2004; Garcia and Cochrane, 2005). Successful implementation of EBM is dependent on access to appropriate quantitative indicators that monitor change in ecosystem components and support analytical assessment to provide evidence-based scientific advice to underpin management (Trenkel and Rochet, 2003; Cury and Christensen, 2005; Jennings, 2005). The need for EBM was formally recognised across most of Europe in 1997 (Heslenfeld and Enserink, 2008) and global acceptance of this need (Garcia 2000, Garcia and Staples, 2000; Jennings, 2004; Garcia and Cochrane, 2005) has subsequently stimulated the development of a proliferation of potential ecological indicators (Greenstreet and Rogers, 2006; Shannon *et al.*, 2010; Shin *et al.*, 2010a; Bundy *et al.*, 2010), often with high levels of redundancy; different indicators reflecting the same ecosystem signal (Blanchard *et al.*, 2010; Greenstreet *et al.*, 2012a; Lyashevska and Farnsworth, 2012).

Marine ecologists providing EBM advice must determine which indicators are most suited to each particular job. This has focused attention on the development of criteria to guide indicator selection (Rice and Rochet, 2005; Rochet and Rice, 2005; Piet *et al.*, 2008; Shin *et al.*, 2010b). In Europe, the Oslo/Paris Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR) was deemed the competent body to supervise development of EBM in the Northeast Atlantic, giving rise to the North Sea Ecological Quality Objectives (EcoQO) pilot study (Heslenfeld and Enserink, 2008; Johnson, 2008). In 2001 OSPAR approached ICES for advice regarding suitable indicators to support potential EcoQOs. In response, ICES first devised a set of seven criteria as a basis for selecting the most appropriate indicators for each EcoQO (ICES, 2001a; ICES, 2001b). For example, in deriving an EcoQO for "fish communities", these criteria suggested that size-based indicators (Blanchard *et al.* 2005; Jennings and Dulvy, 2005; Shin *et al.*, 2005) would be more useful than other types of indicator, e.g. indicators of fish species diversity (Greenstreet, 2008; Greenstreet and Hall 1996; Piet and Jennings, 2005), leading to the development of the "large fish indicator" and the EcoQO "the proportion by weight of fish greater than 40cm in length should be 0.3" (Greenstreet *et al.*, 2011).

Across most European waters, the European Union's Marine Strategy Framework Directive (MSFD) represents the first instance of the application of EBM that covers all human activities in, and pressures on, the marine environment, addresses all major aspects of marine ecosystems, and which is fully underpinned by binding legislation (EC, 2008a). The MSFD lists eleven qualitative Descriptors of Good Environmental Status (GES) and its overarching objective to achieve GES with respect to each Descriptor across seas under European Union (EU) jurisdiction by 2020. The MSFD defines GES as meaning "*the environmental status of marine waters where these provide ecologically diverse and dynamic oceans and seas which are clean, healthy and productive within their intrinsic conditions, and the use of the marine environment is at a level that is sustainable, thus safeguarding the potential for uses and activities by current and future generations, … …*" (EC 2008: paragraph 5 of Article 1). The MSFD places emphasis on the need for action by individual EU Member States (MSs): MSs are required to undertake initial assessments of status; determine a set of characteristics that represent GES; establish appropriate environmental targets and monitoring programmes; and establish and implement a programme of measures to achieve or maintain GES.

However, Article 1 of the MSFD also states that "*Good environmental status shall be determined at the level of the marine region or subregion*" and Article 4 goes on to define the Regions and Subregions explicitly; for example, the North-east Atlantic Ocean, one of four specified Regions, is partitioned into four explicit Subregions: (i) the Greater North Sea including the Kattegat and English Channel, (ii) the Celtic Seas, (iii) the Bay of Biscay and the Iberian Coast, and (iv) the Macaronesian biogeographic region of the Atlantic Ocean, being the waters surrounding the Azores, Madeira and the Canary Islands (EC, 2008a). Article 5 requires MSs sharing marine regions or subregions to co-operate in order to achieve GES at the regional and subregional scale; their marine strategies should be "*coherent and coordinated across the marine region or subregion concerned*" and they should "*endeavour to follow a common approach*" (EC, 2008a). In order to achieve this level of regional/subregional co-ordination, Article 6 suggests that MSs should "*use existing regional institutional cooperation structures, including those under Regional Sea Conventions*" (EC, 2008a). Moreover "*For the purpose of establishing and implementing marine strategies, Member States shall, within each marine region or subregion, make every effort, …… to coordinate their actions with third countries having sovereignty or jurisdiction over waters in the same marine region or subregion*" (EC, 2008a). This confers a specific role, for example, on OSPAR to facilitate the necessary co-operation between Contracting Parties (The European Union, Belgium, Denmark, Finland, France, Germany, Iceland, Ireland, Luxembourg, the Netherlands, Norway, Portugal, Spain, Sweden, Switzerland and the United Kingdom of Great Britain and Northern Ireland) in order to achieve GES across each of the four Subregions that make up the North-east Atlantic Ocean region.

The MSFD timetable is tight: from its adoption in 2008, initial assessments of environmental status in 2012; nomination of indicators and targets also in 2012 (following advice on appropriate indicators published in the 2010 Decision document (EC, 2010)), establishing monitoring programmes to support these indicators in 2014, identifying measures needed to achieve GES by 2015, and their implementation by 2016 in order to achieve GES by 2020 (EC, 2008). Given this ambitious timetable, MSs have focused on meeting their own individual obligations; each MS choosing its own suite of indicators and monitoring programmes to perform initial assessments of status in its own waters (EC, 2014). The need for the co-ordination of the different strategies pursued by each of the MSs that share particular MSFD Regions and Subregions has received less attention, so that the capacity to perform overall assessments of environmental status at a regional seas scale now presents some difficulties for the Regional Seas Conventions.

In 2011 ICES foresaw the possibility of such a situation arising and initiated the development of a set of criteria designed to select the most effective metrics to meet the specific needs of the MSFD indicator functions defined in the Decision document (EC, 2010). In due course ICES were indeed asked by OSPAR to undertake an independent peer review of the technical specifications of the 39 potential "Common Indicators" being considered as the basis for addressing MSFD assessment needs at a Regional/Subregional scale by all MSs sharing Subregions within the North-East Atlantic. These 39 indicators addressed Descriptors 1, 2, 4, and 6, conserving biodiversity, controlling non-indigenous species, maintaining food web structure and function, and maintaining seafloor integrity, respectively. From the perspective of producing a set of "Common Indicators" to be used by all MSs/Contracting Parties to OSPAR across the North-East Atlantic Region, ICES was specifically asked to consider whether the indicators were:

- appropriate to implement at a regional scale;
- currently operational and able to support analytical assessment (ICES, 2013a).

To address this request, a quantitative assessment procedure based around the indicator evaluation criteria was designed so that the effectiveness of each indicator under evaluation could be assessed analytically. A random simulation procedure was then applied so that each of the 39 indicators could be assessed against an *a priori* determined benchmark.

OSPAR had already made some preliminary assessment regarding the extent to which the 39 potential "Common Indicators" were operational or not. Indicators believed to be near operational were already categorised as "common" indicators, whereas indicators deemed to require further development were classed as "candidate" indicators (OSPAR 2013a). Throughout this paper when referring to an OSPAR "Common Indicator", the words are capitalised, but if discussing the categories, "common" or "candidate", lower case font is used.

In this paper we describe the process used to compile the set of criteria used to evaluate the effectiveness of potential OSPAR "Common Indicators". The quantitative evaluation process linked to these criteria is described and the results of a preliminary assessment of 33 potential "Common Indicators" undertaken by the ICES Working Group on Biodiversity Science (WGBIODIV) are presented. WGBIODIV were unable to evaluate six of the potential OSPAR "Common Indicators" (see below for explanation). The benchmark simulation procedure is described and each indicator is assessed against the benchmark agreed by WGBIODIV before commencing their assessment. Examples are presented to demonstrate how the indicator assessment process might be adapted in order to address particular issues deemed to be of critical importance, either by OSPAR, or the MSs involved in carrying out assessments of environmental status in specific regional seas. Finally, we show that quantitative assessment of individual indicators against our set of criteria can be used not only to inform the selection of any particular indicator over another, but also to highlight the particular strengths and weaknesses of any given indicator, and so provide useful insight as to how each indicator might be developed in the future to enhance its effectiveness.

Deriving indicator evaluation criteria

Kershner *et al.* (2011) reviewed 13 published studies, all considering the attributes deemed essential or desirable in an effective "state" indicator, to produce a set of 19 criteria that could be used to guide indicator selection. These were used as a starting point and further refined taking account of specific questions asked by OSPAR of ICES and conclusions drawn from a further nine published studies addressing the subject of indicator selection criteria (ICES 2012; 2013b; 2014). Table 1 lists all 23 published studies that contributed to derivation of the final set of 16 indicator evaluation criteria (Table 2).

Table 1. List of studies considering properties desirable in effective state indicators, and which have influenced the criteria presented here for selecting OSPAR "Common Indicators". Studies numbered 1 to 14 are those originally considered by, and including, Kershner *et al.*, (2011). Studies numbered 15 to 23 are the additional studies taken into account by the ICES Working Group on Biodiversity Science (WGBIODIV).

Number	Reference
1	Doren, R.F., Trexler, J.C., Gottlieb, A.D. and Harwell, M.C. 2009. Ecological indicators for system-wide assessment of the greater everglades ecosystem restoration program. Ecological Indicators, 9: S2–S16.
2	Harwell M.A., Myers V., Young T., Bartuska A., Gassman N., *et al.* 1999. A framework for an ecosystem integrity report card. Bioscience, 49: 543–556.
3	Jackson, L.E., Kurtz, J. and Fisher, W.S 2000. Evaluation guidelines for ecological indicators. EPA/620/R-99/005 US Environmental Protection Agency, Office of Research and Development, Research Triangle Park, NC. 107 p.
4	Jennings, S. 2005. Indicators to support an ecosystem approach to fisheries. Fish and Fisheries 6: 212–232.
5	Jorgensen, S.E., Costanza, R. and Xu, F.L. 2005. Handbook of ecological indicators for assessment of ecosystem health. Boca Raton, FL, USA: CRC Press.
6	Kershner J., Samhouri J.F., James C.A., and Levin P.S. 2011. Selecting Indicator Portfolios for Marine Species and Food Webs: A Puget Sound Case Study. PLoS ONE 6(10): e25248.
7	Kurtz, J.C., Jackson, L.E. and Fisher, W.S. 2001. Strategies for evaluating indicators based on guidelines from the Environmental Protection Agency's Office of Research and Development. Ecological Indicators, 1: 49–60.
8	Landres, P.B., Verner, J. and Thomas, J.W. 1988. Ecological uses of vertebrate indicator species – a critique. Conservation Biology, 2: 316–328.
9	Niemeijer, D. and de Groot, R.S. 2008. A conceptual framework for selecting environmental indicator sets. Ecological Indicators, 8: 14–25.
10	Noss, R.F. 1990. Indicators for monitoring biodiversity: a hierarchical approach. Conservation Biology, 4: 355–364.
11	O'Connor, J.S. and Dewling, R.T. 1986. Indices of marine degradation: their utility. Environmental Management, 10: 335–343.
12	O'Neill, S.M., Bravo, C.F. and Collier, T.K. 2008. Environmental Indicators for the Puget Sound Partnership: A Regional Effort to Select Provisional Indicators (Phase 1). Summary Report. Seattle, WA: National Oceanic and Atmospheric Administration.
13	Rice J.C. 2003. Environmental health indicators. Ocean & Coastal Management 46: 235–259.
14	Rice J.C. and Rochet M.-J. 2005. A framework for selecting a suite of indicators for fisheries management. ICES Journal of Marine Science, 62: 516–527.
15	Beliaef, B. and Pelletier, D. 2011. A general framework for indicator design and use with application to the assessment of coastal water quality and marine protected area management. Ocean & Coastal Management, 54: 84-92.
16	Elliott, M. 2011. Marine science and management means tackling exogenic unmanaged pressures and endogenic managed pressures – A numbered guide. Marine Pollution Bulletin, 62: 651–655.
17	Failing. L., and Gregory, R. 2003. Ten common mistakes in designing biodiversity indicators for forest policy. Journal of Environmental Management, 68: 121–132.
18	Feld, C.K., Sousa, J.P., da Silva, P.M. and Dawson, T.P. 2010 Indicators for biodiversity and ecosystem services: towards an improved framework for ecosystems assessment. Biodiversity Conservation, 19: 2895–2919.

Number	Reference
19	Heink, U. and Kowarik, I. 2010. What criteria should be used to select biodiversity indicators? Biodiversity Conservation, 19: 3769–3797.
20	Painting, S.J., van der Molen, J., Parker, E.R., Coughlan, C., Birchenough, S., Bolam, S., Aldridge, J.N., Forster, R.M. and Greenwood, N. 2013. Development of indicators of ecosystem functioning in a temperate shelf sea: a combined fieldwork and modelling approach. Biogeochemistry, 113: 237-257.
21	Piet, G.J., Jansen, H.M. and Rochet, M-J. 2008. Evaluating potential indicators for an ecosystem approach to fishery management in European waters. ICES Journal of Marine Science, 65: 1449-1455.
22	Rice, J., Arvanitidis, C., Borja, A., Frid, C., Hiddink, J.G., Krause, J., Lorance, P., Ragnarsson, S.A., Sköld, M., Trabucco, B., Enserink, L. and Norkko, A. 2012 Indicators for Sea-floor Integrity under the European Marine Strategy Framework Directive. Ecological Indicators, 12: 174–184.
23	Shin, Y-J., Shannon, L. J., Bundy, A., Coll, M., Aydin, K., Bez, N., Blanchard, J. L., Borges, M. F., Diallo, I., Diaz, E., Heymans, J. J., Hill, L., Johannesen, E., Jouffre, D., Kifani, S., Labrosse, P., Link, J. S., Mackinson, S., Masski, H., Möllmann, C., Neira, S., Ojaveer, H., ould Mohammed Abdallahi, K., Perry, I., Thiao, D., Yemane, D., and Cury, P. M. 2010. Using indicators for evaluating, comparing, and communicating the ecological status of exploited marine ecosystems. 2. Setting the scene. ICES Journal of Marine Science, 67: 692–716.

Table 2. Criteria to evaluate the effectiveness of OSPAR "Common Indicators" intended to support implementation of the MSFD at sub-regional and regional scale. Sixteen criteria are grouped into five main categories. The principle characteristic of indicator effectiveness assessed by each criterion is given. *Importance* weightings, and their associated scores ($I_{criterion}$) (Core = 3, desirable = 2, informative = 1), are shown along with guidelines for assessing the level of *compliance* of an indicator against each criterion, and associated *Compliance* scores ($C_{indicator}$) (1 = fully met, 0.5 = partially met, 0 = not met). Pale blue cells indicate criteria not immediately contributing to the quantitative assessment of an indicator's effectiveness. Criterion 1 identifies whether the indicator being assessed is a "state" or a "pressure" indicator; the compliance score given affects how some of the later criteria in the table are scored, as instructed in the *compliance* guidelines and highlighted where this applies. Criterion 16 is a "tie-breaker" intended to force a choice between high-scoring indicators that essentially fulfil similar ecological functions. The final overall indicator score ($S_{indicator}$) is expressed as a percentage of the total score possible (see text).

Criterion No.	Category	Characteristic	Criterion	*Importance* Weighting	*Importance* Score ($I_{criterion}$)	Guidelines for *Compliance* Assessment and associated *Compliance* Scores ($C_{indicator}$)
1	Type of Indicator	State or pressure	Is indicator a "pressure" indicator being used for want of an appropriate "state" indicator?			Fully met (1): indicator is a "state" indicator; Not met (0): indicator is actually a "pressure" indicator.
2	Quality of underlying data	Existing and ongoing data	Indicators must be supported by current or planned monitoring programmes that provide the data necessary to derive the indicator. Monitoring programmes should have a time series capable of supporting baselines and reference point setting. Data should be collected on multiple sequential occasions using consistent protocols, which account for spatial and temporal heterogeneity.	Core	3	Fully met (1): long-term and ongoing data from which historic reference levels can be derived and past and future trends determined; Partially met (0.5): no baseline information, but ongoing monitoring or historic data available, or historic information available, but monitoring programme discontinued, however potential to re-establish the programme exists; Not met (0): data sources are fragmented, no planned monitoring programme in the future.

Criterion No.	**Category**	**Characteristic**	**Criterion**	***Importance* Weighting**	***Importance* Score ($I_{criterion}$)**	**Guidelines for *Compliance* Assessment and associated *Compliance* Scores ($C_{indicator}$)**
3	Quality of underlying data	Indicators should be tangible	Indicators should be easily and accurately determined using technically feasible and quality assured methods.	Core	3	Fully met (1): data and methods are technically feasible, widely adopted and quality assured in all aspects; Partially met (0.5): potential issues with quality assurance, or methods not widely adopted; Not met (0): indicator is not tangible or doubtful.
4	Quality of underlying data	Quantitative versus qualitative	Quantitative measurements are preferred over qualitative, categorical measurements, which in turn are preferred over expert opinions and professional judgments.	Desirable	2	Fully met (1): all data for the indicator are quantitative; Partially met (0.5): data for the indicator are semi-quantitative or largely qualitative; Not met (0): the indicator is largely based on expert judgement.
5	Quality of underlying data	Relevant spatial coverage	Data should be derived from a large proportion of the MSFD sub-region, at appropriate spatial resolution and sampling design, to which the indicator will apply.	Core	3	Fully met (1): spatially extensive monitoring is undertaken across the sub-region; Partially met (0.5): monitoring does not cover the full sub-region, but is considered adequate to assess status at sub-regional scale; Not met (0): monitoring is undertaken across a limited fraction of the sub-region and considered inadequate to assess status at sub-regional scale.
6	Quality of underlying	Reflects changes in ecosystem	The indicator reflects change in the state of an ecological component	Core	3	**IF CRITERION 1 IS SCORED 0 THEN THE SCORE MUST BE 0.** Otherwise: Fully met

Criterion No.	**Category**	**Characteristic**	**Criterion**	***Importance* Weighting**	***Importance* Score ($I_{criterion}$)**	**Guidelines for *Compliance* Assessment and associated *Compliance* Scores ($C_{indicator}$)**
	data	component that are caused by variation in any specified manageable pressures	that is caused by specific significant manageable pressures (e.g. fishing mortality, habitat destruction). The indicator should therefore respond sensitively to particular changes in pressure. The response should be unambiguous and in a predictable direction, based on theoretical or empirical knowledge, thus reflecting the effect of change in pressure on the ecosystem component in question; signal to noise ratio should be high. Ideally the pressure-state relationship should be defined under both the disturbance and recovery phases.			(1): the indicator is primarily responsive to a single or multiple pressures and ALL the pressure-state[1] relationships are fully understood and defined, both under the disturbance and recovery phases of the relationship. Indicator response should be strongly related to variation in each pressure and the nature of each relationship should be well defined (high signal to noise ratio); Partially met (0.5): the indicator's response to one or more pressures are understood, but the indicator is also likely to be significantly influenced by other non-anthropogenic (e.g. environmental) drivers, and perhaps additional pressures, in a way that is not clearly understood or defined. Correlations between indicator's response and changes in pressure are weaker, indicative of reduced signal to noise ratio. Response under recovery conditions may not be well understood; Not met (0): no clear pressure-state relationship is evident.

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[1] Here the term pressure-state relationship is used in the sense described by Piet *et al.* (2007): e.g. fishing ***pressure*** (fishing mortality rate [F]) – ***state*** of the stock (stock biomass [B]).

Criterion No.	Category	Characteristic	Criterion	*Importance* Weighting	*Importance* Score ($I_{\text{criterion}}$)	Guidelines for *Compliance* Assessment and associated *Compliance* Scores ($C_{\text{indicator}}$)
7	Management	Relevant to MSFD management targets	Clear targets that meet appropriate target criteria (absolute values or trend directions) for the indicator can be specified that reflect management objectives, such as achieving GES.	Desirable	2	Fully met (1): an absolute target value for the indicator is set; Partially met (0.5): no absolute target set for the indicator, but a target trend direction for the indicator is established; Not met (0): targets or trends unknown.
8	Management	Relevant to management measures	Indicator links directly to management response. The relationship between human activity and resulting pressure on the ecological component is clearly understood.	Desirable	2	**IF CRITERION 1 IS SCORED 0 THEN THE SCORE MUST BE 0**. Otherwise: Fully met (1): both response-activity and activity-pressure relationships[2] are well defined - advise can be provided on both the direction AND extent of any change in human activity required and the precise management measures required to achieve this; Partially met (0.5): response-activity and activity pressure relationships are not well understood, or only one of the relationships is defined, but not the other, so that the precise changes in pressure resulting from particular management actions cannot be predicted with

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[2] Here the terms response-activity relationship and activity-pressure relationship are used in the sense described by Piet *et al.* (2007) and Greenstreet *et al.* (2009); e.g. management ***response*** (total allowable catch) – fishing ***activity*** (days-at-sea), and fishing ***activity*** (days-at-sea) – fishing ***pressure*** (fishing mortality rate [*F*]).

Criterion No.	Category	Characteristic	Criterion	*Importance* Weighting	*Importance* Score ($I_{criterion}$)	Guidelines for *Compliance* Assessment and associated *Compliance* Scores ($C_{indicator}$)
						certainty; Not met (0): no clear understanding of either relationship, so that the link between management response and pressure is completely obscure.
9	Management	Comprehensible	Indicators should be interpretable in a way that is easily understandable by policy-makers and other non-scientists (e.g. stakeholders) alike, and the consequences of variation in the indicator should be easy to communicate.	Desirable	2	Fully met (1): the indicator is easy to understand and communicate; Partially met (0.5): a more complex and difficult to understand indicator, but one for which the meaning of change in the indicator value is easy to communicate; Not met (0): the indicator is neither easy to understand or communicable.
10	Management	Established indicator	Indicators used in established management frameworks (e.g. EcoQO indicators) are preferred over novel indicators that perform the same role. Internationally used indicators should have preference over indicators used only at a national level.	Desirable	2	Fully met (1): the indicator is established and used in international policy frameworks; Partially met (0.5): the indicator is established as a national indicator; Not met (0): the indicator has not previously been used in a management framework.
11	Management	Cost-effectiveness	Sampling, measuring, processing, analysing indicator data, and reporting assessment outcomes, should make effective use of lim-	Desirable	2	Fully met (1): little additional costs (no additional sampling is needed); Partially met (0.5): new sampling on already existing programmes is required; Not met (0): new sampling on new monitoring programmes is

Criterion No.	Category	Characteristic	Criterion	*Importance* Weighting	*Importance* Score ($I_{criterion}$)	Guidelines for *Compliance* Assessment and associated *Compliance* Scores ($C_{indicator}$)
			ited financial resources.			necessary.
12	Management	Early warning	Indicators that signal potential future change in an ecosystem attribute before actual harm is indicated are advantageous. These could facilitate preventive management, which could be less costly than restorative management.	Informative	1	**IF CRITERION 1 IS SCORED 0 THEN THE SCORE MUST BE 0**. Otherwise: Fully met (1): indicator provides early warning because of its high sensitivity to a pressure or environmental driver with short response time; Not met (0): relatively insensitive indicator that is slow to respond.
13	Conceptual	Scientific credibility	Scientific, peer-reviewed findings should underpin the assertion that the indicator provides a true representation of variation in the ecosystem attribute in question.	Desirable	2	**IF CRITERION 1 IS SCORED 0 THEN THE SCORE MUST BE 0**. Otherwise: Fully met (1): documented in peer-reviewed literature; Partially met (0.5): documented but not in peer-reviewed literature; Not met (0): not documented, or peer-reviewed literature is contradictory.
14	Conceptual	Metrics relevance to MSFD indicator	For D1 and D6, metrics should fit the indicator function stated in the 2010 MSFD Decision document. This requirement can be relaxed for D4 indicators because the Decision document stipulates the need for indicator development in respect of this Descriptor (but any newly pro-	Core	3	Fully met (1): the metric complies with indicator function; Not met (0): the metric does not comply with indicator function.

Criterion No.	**Category**	**Characteristic**	**Criterion**	***Importance* Weighting**	***Importance* Score ($I_{criterion}$)**	**Guidelines for *Compliance* Assessment and associated *Compliance* Scores ($C_{indicator}$)**
			posed D4 indicators must still fulfil the overall goals stated for D4).			
15	Conceptual	Cross-application	Metrics that are applicable to more than one MSFD indicator are preferable.	Desirable	2	Fully met (1): metric is applicable across several MSFD indicators; Not met (0): no cross-application.
16	Indicator suites	Indicator correlation	Different indicators making up a suite of indicators should each reflect variation in different attributes of the ecosystem component and thus be complementary. Potential correlation between indicators should be avoided.	Desirable	2	Fully met (1): the indicators are uncorrelated; Partially met (0.5): correlation between some indicators; Not met (0): all indicators are correlated.

Deriving a quantitative assessment of indicator effectiveness

Using the criteria as the basis for quantitative assessment of indicator effectiveness required a scoring system. Firstly an "*importance*" weighting was assigned to each criterion; three levels of "*importance*" were considered, "core", "desirable", and "informative", and assigned numerical scores of "3", "2" and "1" respectively. The term "core" was used in preference to the word "essential" because using an "essential" importance weighting might imply the automatic exclusion of any indicator that failed to meet such a criterion. Here the "core" importance weighting means that it is extremely important that the indicator under evaluation meets such criteria, but not absolutely essential. These criteria "*importance*" weightings should be agreed by panels of experts carrying out an assessment of indicator effectiveness before commencing the actual evaluation process. In this instance the WGBIODIV expert panel achieved full consensus on the criteria "*importance*" weightings (Table 2), taking into account views expressed in the 23 published studies (Table 1), the particular needs of the MSFD and OSPAR.

Next a score assessing the extent to which each indicator under evaluation met each criterion had to be assigned. These "*compliance*" scores were assigned independently by each expert taking part in the indicator evaluation process. However, to promote a consistent approach between individual experts, guidelines for determining the level of compliance were devised. Again these were drafted *a priori* of the evaluation process and full consensus was achieved (Table 2). For some criteria, only two levels of "*compliance*" were required, "fully fitted" and "not fitted", but for the majority, the inclusion of a third intermediate level, "partially fitted", was considered necessary. Numerical scores of "1", 0.5" and "0" were assigned to "fully fitted", "partially fitted" and "not fitted" assessments respectively.

The criteria were essentially devised to assess the effectiveness of "state" indicators, but the list of potential "Common Indicators" that OSPAR asked ICES to evaluate also included metrics that were actually indicators of "pressure" (see Table 3). It is inappropriate to evaluate such indicators against criteria related to assessing the sensitivity and responsiveness of "state" indicators to variation in pressure. Any "pressure" indicator should by definition be extremely sensitive and highly responsive to variation in the ecological pressure it purports to measure and would therefore receive an unfairly high score against such criteria. Each potential "Common Indicator" was therefore first assessed against criterion 1, which distinguishes "state" indicators from "pressure" indicators. If the indicator was considered to be a "pressure" indicator, then it was automatically given a *compliance* score of zero against criteria 6, 8, 12, and 13 (highlighted in Table 2), which were considered relevant only to "state" indicators.

Table 3. Abbreviations and categories of the 33 potential OSPAR "Common Indicators" described in the ICG-COBAM Part C: Technical Specifications document available to WGBIODIV. Indicators in the "common" category were deemed by OSPAR capable of being fully operational imminently, while indicators in the "candidate" category were deemed to require some further development. These are the categories pertaining at the time (February 2013) of ICES WGBIODIV's assessment. (P) indicates proposed metrics that are actually indicators of "pressure".

Code	Indicator	Category
Mammals		
M-1	Distributional range and pattern of grey and harbour seal haul-outs and breeding colonies	common
M-2	Distributional range and pattern of cetaceans species regularly present	common
M-3	Abundance of grey and harbour seal at haul-out sites	common
M-4	Abundance at the relevant temporal scale of cetacean species regularly present	common
M-5	Harbour seal and Grey seal pup production	common
M-6	Numbers of individuals within species being bycaught in relation to population	common (P)
Marine birds		
B-1	Species-specific trends in relative abundance of non-breeding and breeding marine bird species	common
B-2	Annual breeding success of kittiwake	common
B-3	Breeding success/failure of marine birds	common
B-4	Non-native/invasive mammal presence on island seabird colonies	common
B-5	Mortality of marine birds from fishing (bycatch) and aquaculture	candidate (P)
B-6	Distributional pattern of breeding and non-breeding marine birds	common (P)
Fish and cephalopods		
FC-1	Population abundance/ biomass of a suite of selected species	common
FC-2	OSPAR EcoQO for proportion of large fish (LFI)	common
FC-3	Mean maximum length of demersal fish and elasmobranchs	common
FC-4	By-catch rates of Chondrichthyes	candidate (P)
Benthic habitat		
BH-1	Typical species composition	common
BH-2	Multi-metric indices	candidate
BH-3	Physical damage of predominant and special habitats	candidate (P)
BH-4	Area of habitat loss	candidate (P)
BH-5	Size-frequency distribution of bivalve or other sensitive/indicator species	candidate
Pelagic habitat		
PH-1	Changes of plankton functional types (life form) index Ratio	common
PH-2	Plankton biomass and/or abundance	common
PH-3	Changes in biodiversity index (s)	common
Food webs		
FW-1	Reproductive success of marine birds in relation to food availability	common
FW-2	Production of phytoplankton	common
FW-3	Size composition in fish communities (LFI)	common
FW-4	Changes in average trophic level of marine predators (cf MTI)	common

Code	Indicator	Category
FW-5	Change of plankton functional types (life form) index Ratio	common
FW-6	Biomass, species composition and spatial distribution of zooplankton	candidate
FW-7	Fish biomass and abundance of dietary functional groups	candidate
FW-8	Changes in average faunal biomass per trophic level	candidate
FW-9	Ecological Network Analysis indicator (e.g. trophic efficiency, flow diversity)	candidate

This decision introduces bias in favour of "state" indicators. Although EBM requires both "state" and "pressure" indicators, so that "pressure-state" relationships can be defined and used as the basis for scientific advice regarding the measures required to achieve GES, assessing the status of different marine ecosystem components and monitoring progress towards GES primarily requires the use of "state" indicators. "Pressure" indicators should only be used for this purpose if no appropriate "state" indicator is available. In a choice between two proposed metrics intended to fulfil the same MSFD indicator role therefore, "state" indicators should be preferentially selected over "pressure" indicators. Although favouring "state" indicators over "pressure" indicators, our approach still allows the effectiveness of both types of indicator to be evaluated together. Situations might arise where a "state" indicator generally scored badly across all criteria but a "pressure" indicator scored particularly well against the criteria applied to it. In such circumstances, the "pressure" indicator could be selected in favour of the "state" indicator if, by virtue of its low score, the "state" indicator is considered to be not fit for purpose. Furthermore, this bias can be taken into account in the benchmarking process.

Criterion 16 considers the level of correlations between indicators and, like criterion 1, is not used in the primary quantitative evaluation of indicator effectiveness. This criterion is intended to ensure that indicators deemed effective actually measure different attributes of an ecosystem component's condition. The purpose of this criterion is to discourage the selection of metrics that perform similar functions; it is essentially a "tie-breaker" to be applied after the main assessment process, designed to force a choice between two high-scoring indicators that perform basically the same function. This should ensure that selected metrics in the eventual suite of indicators provide unique information on ecosystem condition.

For each indicator evaluated, an overall score ($S_{\text{indicator}}$) can be determined by multiplying each criterion "*importance*" weighting score ($I_{\text{criterion}}$) with their respective indicator "*compliance*" scores ($C_{\text{indicator}}$), then summing these products across all evaluation criteria. Dividing this sum by the sum of the criteria "*importance*" weighting scores and multiplying by 100 gives a final indicator score as a percentage of the total score possible:

$$S_{indicator} = \frac{\sum_{criterion2}^{criterion15} I_{criterion} C_{indicator}}{\sum_{criterion2}^{criterion15} I_{criterion}} 100$$

Undertaking an evaluation of potential OSPAR "Common Indicators"

OSPAR (2013a) lists the 39 potential "Common Indicators" ICES was asked to evaluate. When WGBIODIV carried out their evaluation (in February 2013), the working group comprised mainly experts with a detailed knowledge of the North Sea ecosystem. This paper therefore presents a case study covering only the Greater North Sea MSFD Subregion. Two of the indicators related to monitoring the status of non-indigenous species, but too few WGBIODIV members had a working knowledge of marine non-indigenous species for formal evaluation of these indicators to be undertaken. A second document made available to ICES WGBIODIV provided full technical specifications for each indicator (OSPAR 2013b); each expert involved in the indicator assessment based their compliance scores on the evidence presented in this document. No technical specifications were provided in OSPAR (2013b) for four of the "Fish" indicators listed in OSPAR (2013a), so these could not be assessed. Table 3 lists the remaining 33 potential "Common Indicators" that were formally assessed. Each ecosystem component family of indicators was evaluated by at least three experts (four in the case of the food web indicators) with a working knowledge of the ecosystem component in question. These evaluations were carried out independently with no conferring or discussion once underway.

The mean overall assessment score for each indicator, along with the range of overall scores, was determined (Figure 1a). By restricting the evaluation to just a few specific criteria, different aspects of the role of each indicator can be examined explicitly. For example, OSPAR asked ICES to consider whether the indicators could easily be utilised at a regional sea scale. Criterion 5 addresses the spatial coverage aspect of each indicator and the mean evaluation scores against just this criterion are shown in Figure 1b. Similarly, OSPAR asked ICES to consider how easy it might be to make each indicator fully operational. Whether an indicator is operational or not depends on whether it is actually ready for use: has a formulation for the metric been established; is the indicator scientifically sound; are adequate monitoring data, collected across a sufficient area, available to derive the indicator; are these monitoring programmes cost-effective and sustainable; etc? Criteria 2, 3, 4, 5, 10 and 11 (Table 2) relate to these aspects of making indicators operational, and the mean overall score given to each potential OSPAR "Common Indicator" against just these criteria is shown in Figure 1c. Within each ecosystem component family of indicators, there was a strong tendency for "candidate" indicators to be less effective than indicators already tagged as "common" indicators, particularly when assessed against all 14 evaluation criteria, or when assessed against those criteria related to making an indicator operational.

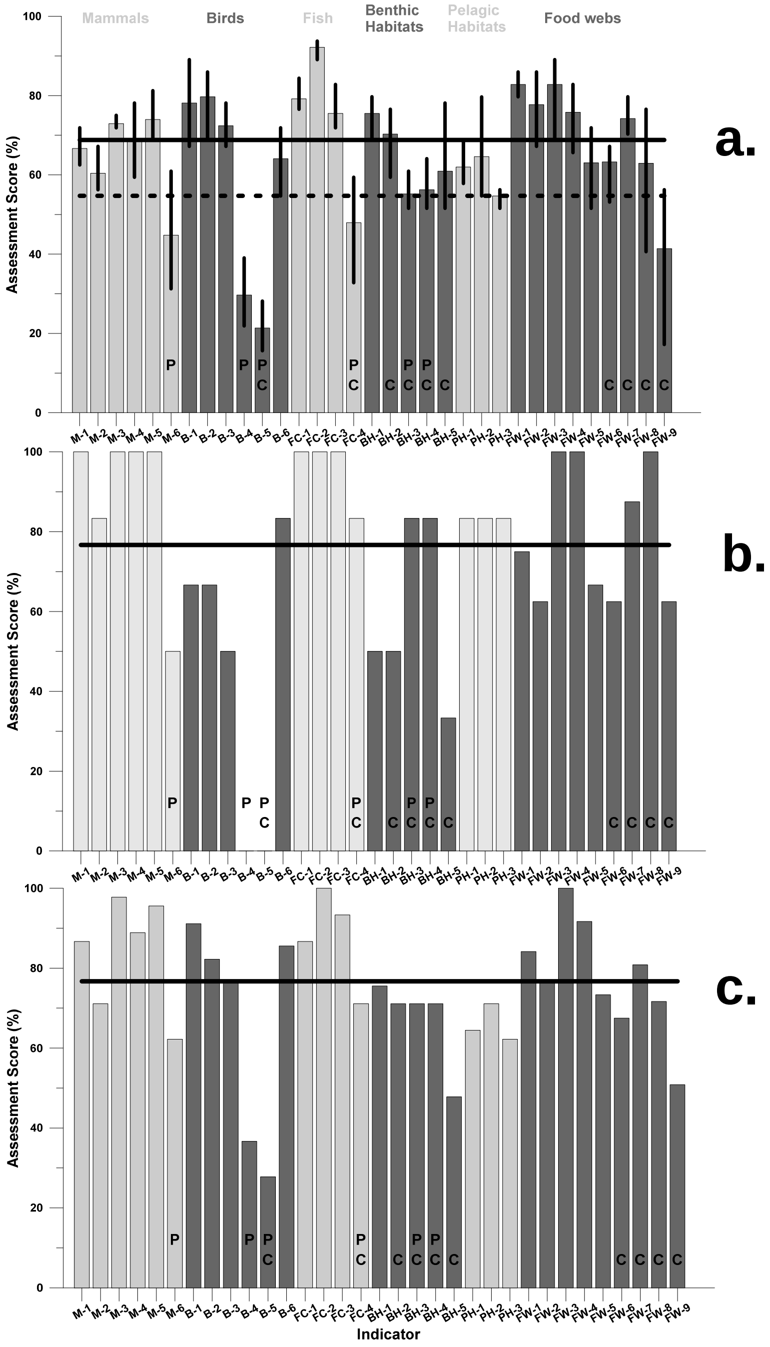

Figure 1. Mean and range of three independent evaluations, four in the case of the food web (FW) indicators) of the effectiveness of 33 potential OSPAR "Common Indicators": a. against criteria 2 to 15 listed in Table 2 for overall indicator effectiveness. ("Pressure" (P) indicators were automatically assigned a zero *compliance* score against four criteria deemed not applicable to "pressure" indicators: see text and Table 2 for further details); b. against criteria 5 listed in Table 2 relevant to spatial coverage (indicators B-4 and B-5 received zero scores from each evaluator); and c. against criteria 2, 3, 4, 5, 10 and 11 listed in Table 2 related to the operationalization of indicators. For abbreviations of indicators see Table 3. Indicators categorised as "candidate" by OSPAR at the time of the evaluation are labelled with "C". The Upper 5 percentile benchmark scores are indicated by the solid black lines for "state" indicators and by the dashed black line for "pressure" indicators (see text and Table 4).

Deriving benchmarks for indicator effectiveness

Having quantitatively assessed each potential OSPAR "Common Indicator" against 14 effectiveness criteria (Table 2), and determined overall scores for each indicator, some form of objective benchmarking procedure was required to interpret these results. Virtual indicator assessments were simulated by randomly assigning one of the possible "*compliance*" scores shown in Table 2 for each criterion used in the assessment, and then determining the overall score across all the criteria used. This randomised simulation process was repeated 100,000 times to determine underlying overall score distributions. The upper 5 percentile of these randomised overall score distributions was used as the benchmark for identifying effective indicators assuming this to provide the best compromise between mistakenly selecting an indicator that was actually ineffective or erroneously rejecting an effective indicator.

Three separate random simulation procedures were undertaken to determine underlying score distributions and establish benchmarks for:

1) effective "state" indicators – using the randomised simulated *compliance* scores for all 14 criteria, numbers 2 to 15 in Table 2;

2) effective "pressure" indicators – using fixed *compliance* scores of zero for criteria 6, 8, 12 and 13, which specifically related to "pressure" indicators, and randomised simulated *compliance* scores for the remaining 10 criteria;

3) ensuring that "state" indicators were operational – using only the randomised simulated *compliance* scores for criteria 2, 3, 4, 5, 10 and 11, which related directly to whether indicators were operational or not (Figure 2).

The means and upper 25, 10 and 5 percentiles of these distributions are summarised in Table 4. Since spatial coverage was assessed by only a single criterion the same simulation procedure could not be used. In this instance, the benchmark threshold of 0.767, used to identify effective near-operational indicators, was adopted; this was also the highest of the four upper 5 percentile benchmark scores.

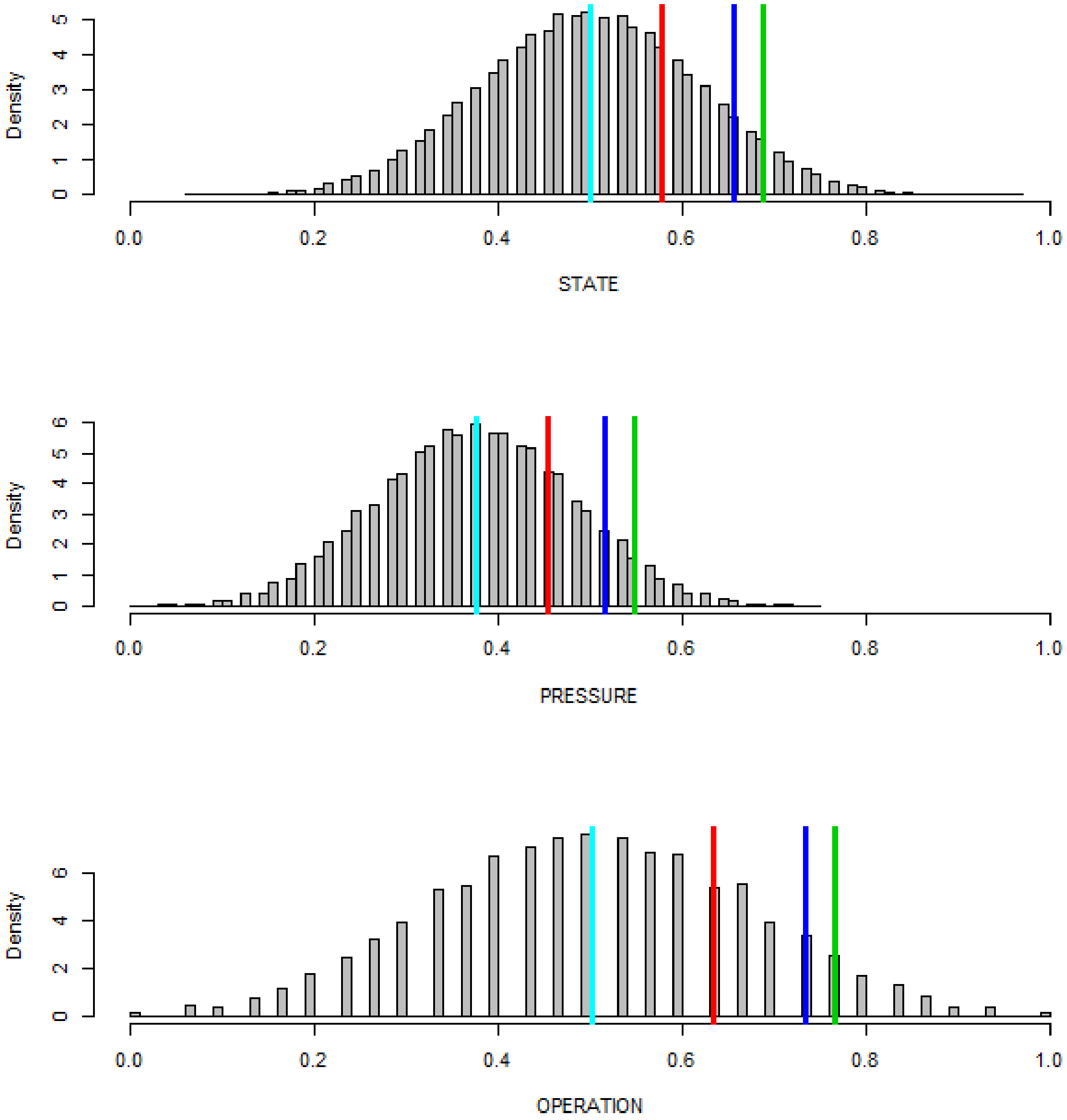

Figure 2. Score distributions for 100 000 virtual indicator assessment simulations, showing mean (light blue), upper 25-percentile (red) upper 10-percentile (blue) and upper 5-percentile (green) scores, computed using randomly sampled indicator "compliance" scores for each criterion: for "state" indicators (top plot); "pressure" indicators (middle plot); and for the subset of criteria considered to reflect indicator operationalization (bottom plot).

Table 4. Summary of the simulated indicator assessments, giving the mean and percentiles (75, 90, 95%) for the distributions of randomised indicator overall scores.

Summary statistic	State	Pressure	Operationalization
Mean	0.500	0.375	0.501
Upper 25 percentile	0.578	0.453	0.633
Upper 10 percentile	0.656	0.516	0.733
Upper 5 percentile	0.688	0.547	0.767

Final indicator assessment

Table 5 summarises the results of all three evaluations, showing which of the potential OSPAR "Common Indicators" exceeded the benchmark thresholds for overall effectiveness, adequate coverage across the Greater North Sea MSFD Sub-region, and ease of operationalization. Only eight indicators met all three benchmarks and seven of these were among the 23 indicators that OSPAR already categorised as "common" indicators. Of these eight indicators, two were marine mammal indicators, three were fish indicators and three were food web indicators. All three of the food web indicators related to, or included, fish components of marine food webs. These results therefore perhaps reflect the greater availability of appropriate fish community survey data and the relatively long history of developing and using ecological indicators that make use of these data.

Table 5. Evaluation of the overall effectiveness of the OSPAR potential "Common Indicators" against criteria 2 to 15 (Overall), against the six criteria related to operationalization of the indicators (criteria 2, 3,4, 5, 10, and 11) and the one criterion related to spatial coverage (criterion 5) of the indicators. Those indicators meeting all three requirements are also indicated. White cells show where indicators meet the benchmark thresholds and dark grey cells show where indicators do not meet the benchmark thresholds. Indicators marked with an asterisk were categorised by OSPAR as "common" while non-asterisked indicators belonged to the "candidate" category. (P) indicates proposed metrics that are actually indicators of "pressure". For abbreviation of indicators see Table 3.

Potential "Common Indicator"	**Overall**	**Operational**	**Spatial Coverage**	**Meets all 3 Benchmarks**
M-1 *	■	□	□	■
M-2 *	■	■	□	■
M-3 *	□	□	□	□
M-4 *	■	□	□	■
M-5 *	□	□	□	□
M-6 * (P)	■	■	■	■
B-1 *	□	□	■	■
B-2 *	□	□	■	■
B-3 *	□	■	■	■
B-4 * (P)	■	■	■	■
B-5 (P)	■	■	■	■
B-6 *	■	□	□	■
FC-1 *	□	□	□	□

FC-2 *				
FC-3 *				
FC-4 (P)	■	■		■
BH-1 *		■	■	■
BH-2		■	■	■
BH-3 (P)		■		■
BH-4 (P)		■		■
BH-5	■	■	■	■
PH-1 *	■	■		■
PH-2 *	■	■		■
PH-3 *	■	■		■
FW-1 *			■	■
FW-2 *		■	■	■
FW-3 *				
FW-4 *				
FW-5 *	■	■	■	■
FW-6	■	■	■	■
FW-7				
FW-8	■	■		■
FW-9	■	■	■	■
TOTAL	17	14	19	8

Nine of the ten indicators categorised as "candidate" indicators failed to meet the benchmark score for being close to operational, confirming their need for further development. Five of these indicators also failed to meet the benchmark score for spatial coverage, suggesting that expanding the spatial extent of monitoring programmes currently supporting these indicators might be a key element of this development process. Food web indicator FW-7 "Fish biomass and abundance of dietary groups" was categorised as a "candidate" indicator, yet in our evaluation, this indicator met all three benchmarks – it was an effective indicator, could be determined across most of the Greater North Sea, and could be made operational within one year. This suggests that this particular indicator might have been incorrectly categorised, and should have been included among those indicators already given a "common" indicator classification.

Addressing operationalization issues

Scores against the six criteria related to operationalization exceeded the benchmark threshold of 77% in only 14 indicators; scores for 19 indicators fell below the benchmark (Table 5). Instead of just using the assessment process to assess the relative effectiveness of different indicators, it can also be used to highlight where an indicator might have particular weaknesses. Table 6 examines each of the 19 poorly-scoring indicators against each of the six operationalization criteria individually. The table clearly illustrates where each of the 19 indicators were particularly weak in relation to operationalization and highlights where research on each indicator could be best applied to raise its score and potential usefulness.

Table 6. Evaluation of the 19 OSPAR potential "Common Indicators" that did not reach the benchmark threshold for operationalization against each of the individual criteria (Cr.) related to operationalization. The values shown are mean scores for each indicator-criterion cell given by the three experts making the assessment expressed as a percentage of the total score possible. Dark grey cells indicate poorest scores (≤ 50%); light grey cells indicate intermediate scores but still below the single criterion benchmark threshold of 77%; white cells indicate highest scores for an indicator against a particular criterion with a score exceeding the benchmark threshold of 77%. Indicators marked with an asterisk are those already assigned to the "common" indicator category at the time of the evaluation while non-asterisked indicators belonged to the "candidate" category. (P) indicates proposed metrics that are actually indicators of "pressure". For abbreviation of indicators see Table 3.

Indicators	**Criteria**					
	Existing data (Cr. 2)	**Metrics tangible (Cr. 3)**	**Quantitative vs qualitative (Cr. 4)**	**Spatial coverage (Cr. 5)**	**Established Indicator (Cr. 10)**	**Cost effectiveness (Cr. 11)**
M-2*	100	83	67	83	0	33
M-6* (P)	67	50	67	50	100	50
B-3*	100	100	100	50	50	50
B-4* (P)	33	83	83	0	0	17
B-5 (P)	33	50	67	0	0	17
FC-4 (P)	67	83	100	83	33	50
BH-1*	83	67	100	50	83	83
BH-2	83	67	83	50	83	83
BH-3 (P)	50	67	100	83	50	83
BH-4 (P)	50	67	100	83	50	83
BH-5	67	50	67	33	50	50
PH-1*	67	50	83	83	33	67
PH-2*	83	67	83	83	67	33
PH-3*	67	50	100	83	0	67
FW-2*	83	67	100	67	83	33
FW-5*	83	83	100	67	33	67
FW-6	83	83	100	50	50	33
FW-8	100	83	83	100	0	67
FW-9	33	50	83	83	0	83

These 19 least-operational indicators included indicators in all species and habitat components of the biodiversity indicators. Proportionally, birds, benthic, pelagic and food web indicators were the least operational. Benthic and pelagic habitat indicators were generally weak in terms of the availability of existing data and metric tangibility, whereas species group indicators, particularly for birds and mammals, were weak in terms of the cost-effectiveness of their data collection and spatial extent of the monitoring programmes. Criteria related to "cost effectiveness", "relevant spatial coverage" and "using an established indicator" were the weakest scoring across the 19 least-operational indicators. This implies that if these indicators are to be made more operational, further work will need in particular to focus on (1) improving the current cost effectiveness of data collection and (2) extending the spatial scale of existing monitoring programmes.

We anticipated seeing the majority, if not all, of the "candidate" indicators in Table 6, since by virtue of being assigned to the "candidate" category, these indicators were deemed to be in need of further development to be made operational. But what was surprising was seeing so many of the indicators already classified as "common" included here. Two explanations are possible. Firstly, it may indeed be the case that the effort required to make the 10 indicators already classified as "common" indicators operational has been under-stated or under-estimated, and that these indicators are indeed further away from being fully operational than was believed to be the case. Alternatively, the evidence to support their categorisation as "common" indicators, close to being operational, was not sufficiently well presented in the Technical Specification document (OSPAR 2013b) preventing their attaining the score they actually merited in the evaluation.

Discussion

Selection of state indicators to support EBM benefits from the use of pre-determined criteria to assess their effectiveness (Rochet and Rice, 2005; Greenstreet, 2008; Shin *et al.* 2010b). Of the 33 potential OSPAR "Common Indicators" evaluated by WGBIODIV, only eight, two mammal indicators, three fish indicators and three food web indicators, met all three benchmarks for an effective indicator, applicable at the appropriate spatial scale, and close to operational. These eight are clearly eligible for adoption by OSPAR as "Common Indicators" for use by all MSs bordering the Greater North Sea to monitor progress towards GES. Of these eight indicators, seven had already been categorised by OSPAR as "common" indicators and the eighth indicator was possibly misclassified. If desired, other particular aspects of an indicator's effectiveness could also be examined by applying different subsets of criteria; each subset selected for their specific relevance to the aspect of indicator effectiveness in question. For example, one might wish to identify indicators able to provide the strongest support to specific management measures. An evaluation using the subset of criteria 7 to 12 could inform such a selection.

The three effective food web indicators are actually dependent on fish abundance-at-length data, so in reality, six of the eight most effective indicators were derived using the primary data collected on groundfish survey monitoring programmes. Two take-home messages emerge from this. Firstly, bottom-trawl surveys, currently undertaken primarily to support fisheries-management, have an essential role to play within the overall monitoring programme necessary to support MSFD implementation. Secondly, considerable investment might be required to bring monitoring programmes servicing other ecological components up to a standard similar to that of the fisheries monitoring programme. This is likely to be particularly important in respect of the benthic and pelagic habitat components. European groundfish surveys are supported by the EC Data Collection Framework (DCF), which provides a "*framework for the collection, management, and use of data in the fisheries sector and support for scientific advice regarding the Common Fisheries Policy*" (EC, 2008b). Funding support to cover monitoring of non-fish ecosystem components was not accessible under the DCF. However, a fish focused approach no longer meets European marine monitoring needs; the remit of these surveys requires extension to include monitoring directed at the wider marine ecosystem (ICES, 2013c). Consequently, as part of the 2013 reform of the EU common Fisheries Policy, the DCF was subsumed into the European Maritime and Fisheries Fund (EMFF), although extent of funding for MSFD monitoring has still to be determined.

For data-rich ecosystem components (e.g. fish and birds), where "pressure" indicators have been proposed to fulfil the need for "state" indicators, these have tended to score poorly compared with actual "state" indicators proposed for the same ecosystem component. Frequently, linkage between the MSFD indicator functions (EC, 2010) and the metrics proposed by OSPAR to fulfil these functions was tenuous. For example, neither "Non-native/invasive mammal presence on island seabird colonies" (B-4) or "Mortality of marine birds from fishing (bycatch) and aquaculture" (B-5) is a metric of seabird "Population condition" (MSFD indicator function 1.3.1: EC 2010); they do not provide any indication of variation in seabird "population demographic condition". Furthermore, metrics used in this manner may well vary in a way that is counter to the intent underlying the metric's proposal. For example, we assume that the proposed B-5 indicator "Mortality of marine birds from fishing (bycatch) and aquaculture" is intended to reduce the mortality imposed on seabird populations by fisheries and so bring about an improvement in seabird population status; thus a decline in bycatch would be expected, which would be reflected in the target set. However, the relationship $C = qPE$ (where q is the catchability coefficient constant, C is the total catch, E is the amount effort and P actual abundance in the sampled area) is universally accepted in fisheries science (King 1995; Jennings *et al.*, 2001; Reiss *et al.*, 2010), so if seabird abundance (equivalent to P) at sea were to increase then, all else being equal, seabird bycatch (C) in fisheries is also likely to increase. If measures implemented under the MSFD to improve seabird population status are successful, an increase in seabird bycatch is the more likely consequence. Meeting targets for real seabird "state" indicators is likely to mean that logical targets for the bycatch "pressure" indicator will be missed.

Despite clear guidelines for assessing the "*compliance*" scores, inconsistency between individual indicator assessments was evident (Figure 1). Post assessment debriefing suggested that variation in the quality of the evidence provided in the Technical Specifications (OSPAR 2013b) was primarily responsible for this. For many potential "Common Indicators", the evidence provided was detailed and clear; experts assessing these indicators drew similar conclusions. For other indicators the text was less explicit, requiring more interpretation on the part of the experts and resulting in more variable assessment outcomes (Rochet and Rice, 2005; Piet *et al.*, 2008). In some instances the necessary evidence was known to exist but was simply not presented in the Technical Specifications document (OSPAR 2013b), with respect to indicator FC-1 the abundance of a suite of "sensitive" fish species (Greenstreet *et al.* 2012b) for example. Assessment of indicator effectiveness is limited by the quality of evidence presented to the expert panel involved (Piet *et al.*, 2008). In this case study, the Technical Specification document (OSPAR 2013b) was written before the indicator evaluation criteria were compiled. Authors of the Technical Specifications were therefore not aware of the criteria against which their indicators would be assessed. Had this order of events been reversed, the Technical Specifications document (OSPAR 2013b) could have been written with a stronger focus on the ensuing assessment to come; evidence supporting each indicator could have been more compelling, and variation between individual evaluations reduced.

In addition, the Technical Specifications document (OSPAR, 2013b) contained no supporting evidence for four potential fish "Common Indicators" (OSPAR, 2013a), even though such information exists. The "Conservation status of elasmobranchs and demersal bony fish (IUCN)" (FC-5) has been discussed (Dulvy *et al.*, 2006; Le Quesne and Jennings 2012), and evaluated by the ICES WGECO (ICES 2012). Greenstreet *et al.* (2012b) evaluated a metric quantifying the "proportion of mature fish in populations of species sampled in groundfish surveys", which could support "Common Indicator" FC-6. Rindorf and Lewy (2012) have reviewed a range of different metrics that could support "Common Indicators" FC-7 "distributional range of a suite of selected species" and FC-8 "distributional pattern within range of a suite of selected species".

This paper has focused on a very specific worked case study: selecting particular indicators, a suite of OSPAR "Common Indicators", to fulfil a particular task, to meet the MSFD requirement of monitoring progress towards GES at the regional seas spatial scale. As such, the wording of the indicator evaluation criteria has been crafted to match that of the MSFD, and the importance weighting assigned to each criterion has been influenced, not only by individual MSs' obligations under the MSFD, but also by the specific questions asked of ICES by OSPAR, who are concerned with meeting these same obligations, but at a regional seas scale. Our approach therefore complies fully with step one of the framework for selecting indicators to support management suggested by Rice and Rochet (2005), i.e. to determine user needs. However, while our particular case study might be highly specific, the underlying concepts and approach are not; these are generic and could easily be adapted and applied anywhere. The indicator evaluation criteria have been developed by building on the knowledge accumulated through 23 previous studies undertaken in locations all around the world and addressing the needs of a variety of different approaches to EBM. Redrafting the criteria text to suit any other particular situation, whilst still retaining their basic meaning, should be a relatively simple task. Similarly, re-evaluating the "*importance*" weighting scores assigned to each criterion should be straightforward and, in any case, this is a task that we have already stated should be undertaken in advance of any new indicator evaluation.

Rice and Rochet's (2005) framework is generic: it can be applied to select indicators to support EBM in any situation or location. Their step two is to develop a list of candidate indicators – this task was done for us by OSPAR (2013a). Steps three to five are to determine the screening criteria, score the indicators, and summarise the scoring results – these tasks exactly reflect the process reported here. Beyond this though, our approach starts to deviate from Rice and Rochet's (2005) framework. Their sixth and seventh steps are firstly to decide how many indicators are needed, and then make a final selection. At present there is no indication from OSPAR, or within the MSFD documentation (EC, 2008a; 2010) as to whether the number of indicators to be used might be limited or not. In Rice's and Rochet's (2005) framework, if the number of indicators required is limited to a fixed number then this would presumably influence the final selection. However, if, as in our case study, this limit is unspecified, then an alternative basis for making a final selection is required: hence, our use of the benchmarking process. Again this same approach could, and perhaps should, be applied in any other set of circumstances.

Acknowledgements

The work reported here was essentially undertaken by two working groups, the Working Group on Biodiversity Science (WGBIODIV) and Working Group on the Ecosystem Effects of Fishing Activities (WGECO). We are grateful to our colleagues in the ICES

WGECO for fruitful discussions that helped shape this work and for their subsequent constructive comments on earlier interim WGBIODIV reports. We also thank members of the UK's Healthy and Biologically Diverse Seas Evidence Group for their suggestions as to how the criteria and evaluation process could be improved and we are grateful for feedback received from the OSPAR Inter-sessional Correspondence Group on the Coordination of Biodiversity Assessment and Monitoring. We thank Bill Turrell for helpful comments on an earlier draft of this section of the WGBIODIV report.

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2 Implementing the Marine strategy Framework Directive: gaps in indicator coverage

In this chapter, the following Terms of Reference are addressed:

1. For the suite of indicators and targets proposed for Descriptor 1 ("Biological diversity is maintained) by Member States in support of the Marine Strategy Framework Directive, WGBIODIV will:
    - c. Assess the extent to which there are gaps in indicators and target coverage for each of the six major ecosystem components suggested by OSPAR
2. OSPAR request 2013-3: Support to the technical specification and application of OSPAR common indicators under D1, 2, 4, and 6. ICES will be requested to undertake an independent peer review of the technical specifications and proposed operational implementation of the indicators that will be presented. The review should consider, from the perspective of producing a set of common indicators for the OSPAR Region:
    - b. whether the set of indicators is sufficient as a set to understand GES;
    - c. identify any gaps;

Summary

Two types of gap in indicator coverage were examined:

- i) Firstly to determine whether all indicator functions provided in the EC Decisions document were covered by at least one indicator in the list of "Common Indicators" being considered by OSPAR, and if not identify which MSFD indicator functions were not being assessed.
- ii) Secondly to determine whether important components or attributes of marine ecosystems were being ignored in either the list of MSFD indicator types provided in the Decision document, or the set of ecosystem components for which "Common Indicators" were being developed by OSPAR.

It was quite clear that a substantial number of indicator roles suggested in the EC Decision document (EC, 2010) were not covered by the list of proposed OSPAR "Common Indicators" (OSPAR, 2013). However, taking issues of indicator redundancy into consideration, it was also apparent that there was little need for many of these gaps to be filled. A large number of potential ecological gaps were also identified. However, it remains to be seen whether the management needed to achieve the targets set for "Common Indicators" covering the major ecosystem components might also be sufficient to address the majority of issues faced by these less represented ecosystem components.

Introduction

ToR 1 was the multiannual term of reference that WGBIODIV set for itself, ToR 2 constituted part of the two Terms of Reference assigned to WGBIODIV to allow ICES to respond to the request for advice from OSPAR. ToR 1(c) and parts (b) and (c) of ToR 2 essentially overlapped one another; both ToRs are therefore addressed in this single section.

Two types of gap were considered:

i. Gaps in MSFD Decision document indicator coverage. This situation arose where the need for a particular type of indicator had been identified in the MSFD Decision document (EC 2010), but where no OPSPAR "Common Indicator" had been proposed to address this MSFD indicator function.
ii. Gaps in ecosystem component coverage. This situation arose either because particular ecosystem components had not been included in the OSPAR approach, or because particular sub-components within a specified ecosystem component (e.g. inshore/coastal fish communities not adequately sampled by the main groundfish surveys) were not covered by proposed OSPAR "Common Indicators" and targets.

Gaps in coverage of the MSFD Decision indicators

Table 1 generally illustrates the linkage specified in the ICG-COBAM Technical specifications document (OSPAR 2013) between each of the 33 metrics proposed as OSPAR "Common Indicators" and the actual indicators needed to meet MSFD requirements defined in the 2010 Decision document. For the Fish and Pelagic Habitats ecosystem components, some Descriptor 1 OSPAR "Common Indicators" had linkages to MSFD food web indicators specified in the ICG-COBAM document, but for the Bird and Mammal ecosystem components similar linkages were not specified, even where WGBIODIV knew such linkage to exist. So in these instances, these linkages are indicated in Table 1; specifically the linkages for the Birds ecosystem component between 4.2.1 and B-2, and between 4.2.1 and B-3. Cells are "greyed-out" where metrics to fulfil the stipulated MSFD indicator function are not strictly necessary. The table can be used to identify potential indicator gaps by looking across the rows. Within each ecosystem component, ideally at least one OSPAR "Common Indicator" colour-coded linkage should be found, and ideally these should be green; orange-coded cells indicate where WGBIODIV had some issue with the specified linkage.

It is clear that the 33 OSPAR "Common Iindicators" currently proposed and assessed by WGBIODIV do not address all the MSFD indicator functions listed in the 2010 Decision document.

i. Within the three species-level ecosystem components, no OSPAR "Common Indicator" addresses the requirement for indicators of "population genetic structure" (MSFD indicator 1.3.2).
ii. No Seabird or Mammal ecosystem component OSPAR "Common Indicator" addresses the requirement for indicators of "composition and relative proportions of ecosystem components" (MSFD indicator 1.7.1).
iii. None of the OSPAR "Common Indicators" addressing MSFD food web indicator needs (4.1.1, 4.2.1, and 4.3.1) appear to utilise benthic invertebrate metrics.
iv. MSFD indicator 1.1.3 applies specifically to benthic species and habitats, yet none of the Benthic Habitat OSPAR "Common Indicators" link to this MSFD indicator.
v. No OSPAR "Common Indicator" for Fish and Cephalopods is linked to the two Descriptor 1 distribution range and distribution pattern MSFD indicators (1.1.1

and 1.1.2), or to the population demographics MSFD indicator (1.3.1). WGBIODIV notes, however, that two distribution indicators (FC-7 and FC-8) and a proportion of mature fish (FC-6) indicator were listed in part B of ICG-COBAM's report as potential "Candidate Indicators", but no details were provided in the Technical Specifications Part C. WGBIODIV was aware of such developments (e.g. Greenstreet *et al.*, 2012a).

vi. None of the Benthic Habitat OSPAR "Common Indicators" addressed the habitat-level Descriptor 1 MSFD indicator requirements for distributional range (1.4.1), distributional pattern (1.4.2), or habitat volume (1.5.2).

Table 1. The EC MSFD Decision indicators to which OSPAR "Common Indicators" (white column headings) and OSPAR "Candidate Indicators" (grey column headings) are linked in the ICG-COBAM Technical Specification document. Grey-filled cells indicate where no linkage in required. Linkages specified in the ICG-COBAM document are indicated by green-filled cells (bright green for OSPAR "Common Indicators" and pale green for OSPAR "Candidate Indicators") where WGBIODIV had no issues, or by orange-filled cells (again bright orange for OSPAR "Common Indicators" and pale orange for OSPAR "Candidate Indicators") where WGBIODIV had issues. These issues are numerically coded: (1) The OSPAR "Common Indicator" does not seem to relate to the MSFD indicator; (2) This multi-metric indicator does not fit the definition of the MSFD indicator, but is implicitly part of it; (3) This OSPAR "Common Indicator" is considered relevant by OSPAR, but is not a MSFD indicator specified in the Decision document; (4) WGBIODIV considered this linkage to be tenuous, with insufficient detail provided in the ICG-COBAM Technical Specifications document to be convincing.

EC indicator	M-1 (range pattern grey & harbour seals)	M-2 (range pattern cetaceans)	M-3 (abundance grey & harbour seals)	M-4 (abundance cetaceans)	M-5 (seal pup production)	M-6 (bycatch)	B-1 (trends in species)	B-2 (breeding success kitiwake)	B-3 (breeding success/failure marine birds)	B-4 (mammals on seabird colonies)	B-5 (bycatch)	B-6 (distribution)	FC-1 (abundance/biomass)	FC-2 (LFI)	FC-3 (mean max length)	FC-4 (bycatch rate of Chondrichthyes)	BH-1 (typical species)	BH-2 (multimetric indices)	BH-3 (damage)	BH-4 (area habitat loss)	BH-5 (size-freq sensitive species)	PH-1 (changes functional types)	PH-2 (biomass/abundance)	PH-3 (changes biodiv index)	FW-1 (repr success in relation to food availability)	FW-2 (production of phytoplankton)	FW-3 (LFI)	FW-4 (change in average trophic level of predators)	FW-5 (change funct groups plankton)	FW-6 (zooplankton)	FW-7 (fish biomass and dietary functional groups)	FW-8 (Biomass Trophic Spectrum)	FW-9 (Ecological Network Analysis)
Biodiversity	MAMMALS						BIRDS						FISH & CEPH				BENTHIC HABITATS					PEL. HAB.			FOOD WEBS								
1.1.1 Distributional range	🟩	🟩															▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨
1.1.2 Distribution pattern within the latter	🟩	🟩										🟩					▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨
1.1.3 Area covered by the species	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨						▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨
1.2.1 Abundance and/or biomass			🟩	🟩			🟩						🟩			(1)	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨
1.3.1 Population demographics					🟩	(1)		🟩	🟩	(1)	(1)						▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨
1.3.2 Population genetic structure																	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨
1.4.1 Distributional range	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨						🟩			▨	▨	▨	▨	▨	▨	▨	▨	▨
1.4.2 Distributional pattern	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨						🟩			▨	▨	▨	▨	▨	▨	▨	▨	▨
1.5.1 Habitat area	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨				▩					▨	▨	▨	▨	▨	▨	▨	▨	▨
1.5.2 Habitat volume	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨									▨	▨	▨	▨	▨	▨	▨	▨	▨
1.6.1 Condition of the typical species and communities	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	(1)	▨		(2)				🟩		🟩	▨	▨	▨	▨	▨	▨	▨	▨	▨
1.6.2 Relative abundance and/or biomass	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	(1)	▨	▨		(2)					🟩		▨	▨	▨	▨	▨	▨	▨	▨	▨
1.6.3 Physical, hydrological and chemical conditions	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨			▩						▨	▨	▨	▨	▨	▨	▨	▨	▨
1.7.1 Composition and relative proportions of ecosystem components (habitats and														🟩	🟩							🟩		🟩	▨	▨	▨	▨	▨	▨	▨	▨	▨
Food webs																																	
4.1.1 Performance of key predator species																									🟩	🟩							
4.2.1 Large fish (by weight)								🟩	🟩					🟩													🟩						
4.3.1 Abundance trends of functionally important selected groups/species.																						🟩	🟩				🟩	🟩	🟩	▩	▩	▩	(3)
Seabed integrity																																	
6.1.1 Type, biomass and areal extent of relevant biogenic substrate.	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨			▩			▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨
6.1.2 Extent of the seabed significantly affected by human activities.	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨			▩	▩		▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨
6.2.1 Presence of particularly sensitive and/or tolerant species.	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨		(2)				▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨
6.2.2 Multi-metric indexes assessing benthic community condition …	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨		🟩				(4)	▨	▨	▨	▨	▨	▨	(4)	▨	▨	▨	▨
6.2.3 Proportion of biomass or number of individuals .. above some length/size.	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	🟩				▩	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨
6.2.4 Parameters describing the...size spectrum of the benthic community.	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨						▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨

The immediate question therefore is "Are metrics that perform all of the indicator functions listed in the 2010 Decision document really required?" In some instances the answer appears obvious. It seemed inconceivable to WGBIODIV that any assessment of the state of a marine food web could reach a convincing conclusion of GES where no metrics relaying the contribution of benthic invertebrates to food web structure and functioning had been included in the analysis. Benthic invertebrates constitute a critical food resource for a large fraction of the fish community (Greenstreet *et al.*, 1997; Heath, 2005) and the benthos plays a key role in important food web processes such as resource recycling. In other instances, apparent MSFD indicator gaps may not be so critical.

For each of the Descriptors, D1, D4 and D6, the EC Decision document lists what is essentially a suite of indicators intended to cover key aspects of relevance to each Descriptor in respect of each of the species and habitat ecosystem components. But where suites of indicators are used to monitor change in the status of different attributes of any particular marine ecosystem component, care is required to minimise or eliminate indicator redundancy, or at the very least to take account of this (Blanchard *et al.*, 2010; Bundy *et al.*, 2010; Greenstreet *et al.*, 2012b). Indicator redundancy occurs when two or more indicators in a suite effectively reflect change in a single attribute of an ecosystem component, so that variation in the two metrics is correlated.

Indicator redundancy is undesirable because it can generate misleading impressions of actual progress towards GES. Consider five metrics, four of which reflect change in one particular ecosystem component attribute and are therefore correlated, while the fifth reflects change in a second ecosystem component attribute and so is uncorrelated with the other four metrics. Only two attributes are actually monitored. If GES is achieved for one, but not for the other, we might then consider the job to be 50% done. However, this is not the impression that would be gained from any assessment of the five metrics. Depending on which attribute was at GES and which attribute was reflected by the different metrics, we would appear to be either 20% or 80% of the way towards GES.

If filling some of the gaps evident in Table 1 resulted in increased indicator redundancy, i.e. the inclusion of indicators that correlated closely with one another, this could be deemed undesirable, so filling such gaps might not be necessary. For example, given the spatial sampling designs of many groundfish surveys capable of providing data to assess the biodiversity status of fish communities, it is perhaps surprising that no "Common Indicators" have been proposed by OSPAR to fulfil the distributional range (MSFD indicator 1.1.1) or distributional pattern (MSFD indicator 1.1.2) indicator functions. The abundance-occupancy relationship (i.e. the more abundant a species is, the wider its distribution) is a general ecological pattern, applying both within species (as a species increases in abundance, so its range increases) and between species (more abundant species occupy larger ranges than rare species) (Gaston *et al.*, 2000; Blackburn *et al.*, 2006). Density-dependent habitat selection causes species to expand into more marginal habitats as abundance increases, or to contract into the most optimal habitat as abundance declines (Fretwell and Lucas, 1970; Partridge, 1978; MacCall, 1990). Both intra- and inter-specific abundance-occupancy relationships have been demonstrated in fish populations (Overholtz, 2002; Hinz *et al.*, 2003; Fisher and Frank, 2004; Blanchard *et al.*, 2005). There is therefore a strong theoretical basis for believing that metrics of demersal fish population distributional range, distributional pattern within range and abundance should all covary in a predictable way, implying high levels of redundancy among them. Given that

ICG-COBAM propose metrics of fish population abundance, additional metrics of distributional range and distributional pattern within the range may in reality impart little further insight regarding actual variation in fish biodiversity; filling these MSFD indicator gaps may not be necessary.

The 2010 Decision document is quite explicit that MSFD indicators 1.4.1, 1.4.2, 1.5.1, 1.5.2, 1.6.1, 1.6.2, and 1.6.3 all relate to the "habitat-level of biodiversity", and so are pertinent to both the Benthic Habitats and Pelagic Habitats ecosystem components, but not to Birds, Mammals and Fish, considered by OSPAR to be "species-level of biodiversity" components. Despite this the ICG-COBAM Technical Specification document linked two Fish component OSPAR "Common Indicators", the Large Fish Indicator (FC-2) and the mean maximum length (FC-3), to MSFD indicator types 1.6.2 and 1.6.1 respectively. WGBIODIV considered this to be erroneous and confusing, and that both OSPAR "Common Indicators" were more appropriately linked to the ecosystem/community level of biodiversity covered by MSFD indicator 1.7.1; a linkage also specified in the ICG-COBAM Technical Specification document. WGBIODIV would suggest that the linkages between FC-2 and 1.6.2, and between FC-3 and 1.6.1 be removed so that a consistent underlying rationale linking the OSPAR "Common Indicators" to the MSFD indicators is maintained across all ecosystem components.

Ecological gaps

The proposed OSPAR "Common Indicators" reflect the six OSPAR ecosystem components (seabirds, marine mammals and reptiles, fish & cephalopods, benthic habitats, pelagic water column habitats, and food webs). However, focus on such broad groups may mean that more "minor" ecosystem components are ignored, and some of these may indeed have a bearing on determining whether GES at the whole ecosystem level has actually been achieved in respect of any given OSPAR region. WGBIODIV therefore considered a list of potential ecological gaps, which may aid the further development of new indicators and identify potential additional monitoring needs.

Genetic diversity

None of the OSPAR "common indicators" address genetic diversity. This is likely due to a lack of existing information and the potentially high cost of genetic analysis.

Coastal and inshore fish communities

Coastal fish communities are not monitored under the WFD. New monitoring programmes may be required to cover this apparent gap between MSFD and WFD. Just as the shelf-sea surveys venture rarely into deeper water regions, neither do they sample in shallower coastal zone waters. Consequently, none of the indicators and monitoring programmes currently under consideration address the biodiversity of inshore and coastal fish communities. Shallow coastal waters are important during the juvenile phase of the life-history of many fish species. This could be a potential gap that may therefore need to be addressed.

Sensitive species (including threatened and declining)

Many species of skates, rays and sharks are of conservation concern and further indicator development to address sensitive species may be necessary. Two OSPAR "Candidate

Indicators" were proposed in relation to sensitive species: FC-4 and FC-5. FC-4 is a pressure indicator and FC-5 was not described in the ICG-COBAM Technical Specifications (Part C). Although many sensitive species, such as Chondrichthyan fishes, are not well sampled by current surveys, for those species that are reliably sampled the FC-1 indicator could be developed as shown by Greenstreet *et al.* (2012a).

Rocky habitats

The sampling of rocky habitats for the status assessment of benthic fauna and fish species may not be adequate. Fisheries surveys generally do not sample these habitats because of the risk of sustaining damage to the trawl. Rocky habitats often host sensitive species and may provide refuge from fishing (Shephard *et al.*, 2012). Rocky habitats could be monitored by non-invasive or non-lethal sampling techniques, such as scuba diving, image-based technologies, traps or pots.

Deep water habitats and species

Pressures on benthic habitats tend to be more evident in coastal areas so these have traditionally received more attention. However, impacts of anthropogenic activity on deep sea habitats have been identified, for example in relation to deep sea corals (Hall-Spencer *et al.*, 2002). Indicators reflecting variation in the status of deep water sponges and corals may be necessary, potentially requiring new monitoring programmes. The majority of groundfish surveys operate in coastal shelf seas. For example, one of the most extensive surveys, the North Sea IBTS, does not survey the deeper parts of the North Sea, Skagerrak and Kattegat. Some deep water fish survey work is undertaken off the shelf edge to the west of the United Kingdom and Bay of Biscay, but it needs to be established that these surveys are adequate to support monitoring of the status of deep water fish communities.

Highly migratory species

Highly migratory species (including sharks, tunas or marine reptiles) are only partly addressed by the OSPAR "Common Indicators", but for many species adequate monitoring programmes are lacking. Large and fast swimming species such as basking shark (*Cetorhinus maximus*) are monitored on an *ad-hoc* basis by visual observation on some scientific fishing surveys. This may require additional monitoring effort. None of the OSPAR "Common Indicators" considers the status of marine reptiles. Leatherback turtle (*Dermochelys coriacea*) is known to occur frequently in the OSPAR regions III and IV (OSPAR, 2009). An abundance indicator for this species may be appropriate. Diadromous fish (salmon, eels) are represented in other Directives (i.e. the WFD) which cover the freshwater stages of their life history. However, a low survival rate during the marine phase of their life-cycle is known to be an issue in some species and this is not monitored by the OSPAR "Common Indicators" proposed. While some of the issues in relation to marine survival are likely to occur outside the area covered by the MSFD, interactions with other ecosystem components and management measures may occur.

Microplankton & microbenthos

For PH-1 and FW-5, OSPAR identified micro-, pico-, nano-phytoplankton and bacteria and micro-zooplankton including ciliates as being under-sampled. Considering that these groups are essential components of the microbial loop of marine food webs, monitoring and assessing their status may be important.

Cephalopods

Cephalopods are caught in fisheries research surveys (e.g. IBTS) and hence data for the status assessment of at least some species under FC-1, FC-7 and FC-8 should be available. Fishing pressure on cephalopods in the North Sea is relatively low and cephalopod populations might be assumed to be at GES in this region. In other OSPAR regions, however, new targeted monitoring for cephalopods (e.g. octopuses) may be necessary.

Locally extirpated species

In certain regions there may be efforts to re-establish locally extinct species (e.g. sturgeon *Acipenser sturio*). Specified indicators and monitoring programs may help evaluate the success of these measures.

Reproduction of cetaceans

Similar to assessing the productivity of grey seals, the productivity of cetaceans could also be assessed if the data were available. For example, for some species such as harbour porpoises (*Phocoena phocoena*) and some species of dolphins it may be possible to assess the proportion of calves within local populations.

Habitat use of seals

Spatial use by seals across the wider marine habitat is not considered by the OSPAR mammal indicators. Activity patterns of seals may be important to assess spatial conflicts with human activities such as passive fishing or construction of offshore wind parks.

Concluding comments on ecological gaps

Just as it may not be necessary to identify metrics to perform all the MSFD indicator functions, it is also unlikely to be essential that absolutely every component of marine ecosystems needs to be monitored in order to confirm that, at a holistic ecosystem level and at sub-regional or regional scale, GES has been achieved. Identification of the critical ecosystem components, and the most effective indicators to monitor these, is what is required. For example, in the Greater North Sea cephalopods are largely caught and landed as a bycatch. If fisheries were to be managed such that D3 targets for fishing at maximum sustainable yield were met, and this resulted in GES being achieved for the majority of fish community D1 and D4 indicators, particularly those relating to fish species with similar life history characteristics and similar food web functional roles as cephalopods, could we not then simply assume that cephalopod populations would be unlikely to be perturbed by human activity to an unsustainable degree? Would a separate suite of D1 and D4 cephalopod indicators, along with defined GES targets and the monitoring programmes needed to support these, really be necessary?

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3 Effects of climate change on marine ecosystems, and associated challenges for setting management targets for ecological indicators

In this chapter, the following Terms of Reference are addressed:

1. For the suite of indicators and targets proposed for Descriptor 1 ("Biological diversity is maintained) by Member States in support of the Marine Strategy Framework Directive, WGBIODIV will:
    a. Consider the consequences of recent changes in environmental conditions associated with climate change on the relevance of targets set on the basis of historical data baselines.

Summary

The European Union's (EU) Marine Strategy Framework Directive (MSFD) requires 'good environmental status' (GES) by 2020. Implementing the MSFD involves the use of a large number of indicators to monitor change in various attributes of a variety of different ecosystem components. For each indicator, targets representing the desired indicator value must be set: the value of the indicator expected when the attribute of the ecosystem component in question would be deemed to be at GES. However, the MSFD is cognizant of the fact that the marine environment is not a static entity, and that variation in environmental conditions can affect the state of these ecosystem component attributes. GES for each ecosystem component attribute could therefore be a moving target reflecting changes in prevailing environmental conditions.

In this section, long-term changes in the marine environment of the Northeast Atlantic are reviewed and possible future trends in environmental variables and a number of different climate change scenarios are presented. Considerable variation in both water temperature and salinity has occurred across the region during the course of the 20th century and into the early 21st century with clear periods of lower or higher than average temperature apparent, generally associated with either reduced or enhanced salinity. These phase changes have been linked to distinct periodic events, such as the "Great Salinity Anomaly of the 1970s, or related to cyclical changes such as the "North Atlantic Oscillation" or the "Atlantic Multi-decadal Oscillation". Recent warming of waters of the Northeast Atlantic region has been associated with global climate change linked to anthropogenic use of fossil fuels leading to build up in carbon dioxide and other 'greenhouse' gasses in the earth's atmosphere. Current climate change scenarios suggest that the world's oceans, including the Northeast Atlantic, are likely to continue along a warming trend at least for the next thirty years or more.

These changes in environmental conditions have had marked effects throughout the marine ecosystems of the Northeast Atlantic. Changes related to variation in temperature and salinity among phytoplankton, zooplankton, benthic invertebrates, fish, seabird and marine mammal communities have all been observed and these are reviewed. Based on the historical changes observed, projected effects of a continuing increase in sea temperature are considered and the implications for current targets based on historical or current reference conditions for particular plankton, benthic invertebrate, fish, seabird and ma-

rine mammal indicators are assessed. Several targets may not adequately account for recent and future changes in the marine environment associated with ongoing climate change. Consequently, there is a real risk that the indicators involved may fail to meet these targets. Such failures could give the impression that management has been inadequate to achieve the goals of the MSFD. However, the alternative interpretation, that targets set for the failing indicators were inappropriate because they failed to take changes in the marine environment into account adequately and therefore did not reflect actual environmental conditions prevailing at the time of the assessment, should first be considered and discounted before introducing even more stringent management action.

Introduction

The Marine Strategy Framework Directive (MSFD) requires that good environmental status (GES) be achieved in European Seas for a broad range of ecosystem components by 2020 (EC, 2008). Key attributes of each ecosystem component therefore require formal assessment. There are three principal elements to such assessments (Link, 2005). Establishing over-arching goals – identifying the goods and services society wants from marine ecosystems – is the first element. These over-arching goals are already expressed in the MSFD itself: the maintenance of biodiversity, the intention of providing diverse and dynamic oceans and seas that are clean healthy and productive, and the aim of exploiting marine natural resources in a sustainable way so that their long-term productivity is safeguarded. This leaves the last two elements – the monitoring of ecosystem components and the setting of targets consistent with management goals – to be addressed here. Firstly, change in the "condition" of ecosystem component attributes identified as being important must be monitored using appropriate "state" indicators (Trenkel and Rochet, 2003; Jennings 2005, Shannon *et al.*, 2010; Shin *et al.*, 2010a; EC, 2010), chosen on the basis of their performance against specified selection criteria (Rochet and Rice, 2005; Piet *et al.*, 2008; Greenstreet *et al.*, submitted). Secondly, the current condition of each ecosystem component attribute has to be compared with a target condition: an indicator value consistent with 'favourable status' (Rice 2003; Sainsbury and Sumaila, 2003; Jennings, 2005; Rice, 2011), or in MSFD terms, an indicator value that represents GES for the ecosystem component attribute in question. These target values need to be established *a priori* of the assessment. Since achieving GES across European seas lies at the heart of the MSFD, considerable emphasis is placed on the target-setting aspect of marine ecosystem assessment.

To date, two broad approaches to setting targets have been employed, an "empirical" approach and a "modelling" approach; both can be exemplified by the way that targets have been set for the large fish indicator (LFI) in different marine regions. In the North Sea, the early 1980s was the last time when ICES advice was to maintain exploitation levels at *status quo* (ICES 2006a), implying that at this time, although heavily exploited, fish stocks were still deemed to be in acceptable condition. The early 1980s was therefore considered a suitable reference period for defining the "sustainable use state" so that LFI value of 0.3 observed at this time was adopted as the management target. In the southern Bay of Biscay, suitable groundfish survey data to determine a LFI value at a time when stocks were deemed to be sustainably exploited were not available. The linear regression relationship between fishing mortality and the LFI was therefore used to model the LFI value expected, should stocks in the area once again be exploited at sustainable levels (Modica *et al.*, 2014). Since the MSFD defines GES as being commensurate with sustainable use of natural resources (EC, 2008; Jennings and Le Quesne 2012), the approach used in each study, and their associated LFI targets, remains perfectly valid for defining targets consistent with sustainable use, and therefore for demersal fish communities at GES. The empirical approach has been used most frequently for setting targets for indicators used to assess the "state" of ecosystem components beyond the main commercial fish species (Greenstreet and Rogers, 2006). For fisheries management both approaches have been used (Box 1). Shin *et al.* (2010) suggest a third approach to target setting, the use of "expert judgement panels" to decide when state is unacceptable. However, proposed targets would still reflect the experience of those involved and, depending on the relative composition "theoretical" and "empirical" scientists within each expert panel, are likely to be an amalgamation of the two approaches.

Spawning stock biomass (SSB) is the "state" indicator used to monitor change in the condition of stocks and the precautionary management reference point (B_{pa}) is determined empirically. Trends in SSB and recruitment are examined to establish the SSB below which recruitment becomes impaired, the limit reference point (B_{lim}). Should SSB < B_{lim}, emergency remedial management measures to minimise fishing mortality and promote stock recovery would be triggered. B_{pa} takes account of imprecision in estimating *SSB*, incorporating sufficient buffer to ensure that so long as B_{pa} is not breached, there is minimal likelihood of SSB < B_{lim}. Fishing mortality (F) is the "pressure" indicator used in the indicator-based "Pressure"-"State"-"Response" (PSR) fisheries-management framework (Garcia and Staples, 2000; Jennings, 2004). Population dynamics models (e.g. Shepherd, 1999) are used to estimate limit reference points for fishing mortality (F_{lim}), the maximum sustainable fishing mortality rate that still retains SSB > B_{lim}. Again, to allow for imprecision in estimating both SSB and F, a precautionary fishing mortality limit (F_{pa}) consistent with ensuring that F < F_{lim} and maintaining SSB > B_{pa} is used (Rice, 2009). With regard to implementation of the MSFD, fisheries managers are being asked to achieve maximum sustainable yield (MSY) and again the same population dynamics models are being used to determine appropriate reference points related to these new objectives, such as F_{MSY}.

Box 1. Use of both the empirical and the modelling approaches to set targets for fisheries management.

Even though fishing has widely been presumed to be the primary driver of change in marine ecosystems (Watling and Norse, 1998; Jennings and Kaiser, 1998; Jackson *et al.*, 2001; Hilborn *et al.*, 2003; Myers and Worm, 2003; Worm *et al.*, 2006), environmental drivers also have a profound influence (Link *et al.*, 2002; Stenthseth *et al.*, 2002; Blanchard *et al.*, 2005; Frank *et al.*, 2007; Edwards *et al.*, 2010). The marine environment of the northeast Atlantic has varied markedly over the last 100y (Reid and Valdes, 2011), and there is considerable evidence that this has had marked consequences for marine ecosystems (Beaugrand *et al.*, 2009; Rijnsdorp, *et al.*, 2010; Hoegh-Giulderg and Bruno 2010; Heath *et al.*, 2012; Frederickson *et al.*, 2013). Even relatively small changes in sub-regional scale average temperature can trigger habitat-switching where species close to their temperature limits exist at the edge of their biogeographic range (Edwards *et al.*, 2013). This highlights a potential problem with both approaches to target setting. The empirical method draws on indicator values observed during a "reference period" at some time-point in the past. However, in setting these "reference period" indicator values as the target, the underlying assumption is that only human pressure on the ecosystem has changed, so that if this pressure can be managed appropriately, these same historic indicator values could once again be achieved. Similarly, models used to predict indicator values under different levels of human pressure are invariably parameterised using previously collected data. Neither approach takes account of intervening change in the marine environment. Previous historic states might simply be biologically unattainable under currently prevailing environmental conditions and model parameter values based on historic data may not be pertinent in a changed marine environment.

The MSFD acknowledges that the marine environment is not a static entity; subject to environmental drivers, marine ecosystems therefore also change over time. In stating "*In view of the dynamic nature of marine ecosystems and their natural variability, and given that the pressures and impacts on them may vary with the evolvement of different patterns of human activity and the impact of climate change, it is essential to recognise that the determination of good environmental status may have to be adapted over time. … …* " (EC, 2008), the MSFD explicitly acknowledges that GES will not be a constant state and will periodically have to be reassessed. This theme is reprised in the Decision document (EC, 2010), which states that GES for Descriptor[3] 1: "*Biological diversity is maintained*" will be achieved when "*The quality and occurrence of habitats and the distribution and abundance of species are in line with prevailing physiographic, geographic and climate conditions*" (EC, 2010). Again this infers that as prevailing physiographic, geographic and climate conditions vary, so too will the quality and occurrence of habitats and the distribution and abundance of species. This clearly implies that where targets representing GES for indicators used to monitor change in particular ecosystem components have been determined on a historical basis, these may need to be reassessed and revised to take account of intervening environmental change.

As MSFD monitoring and assessment progresses, some indicator time-series trajectories may fail to converge on current targets, and in such circumstances it might be tempting to presume that management measures have been inadequate. At this point it is important to keep in mind that the MSFD is not just legislation intended to promote marine conservation, it is also intend to promote the adoption of an "*ecosystem-based approach to the management of human activities while enabling a sustainable use of marine goods and services*" (EC, 2008). If the response to missing targets is always to impose more restrictive management, then this will certainly compromise capacity to exploit marine natural resources fully. In circumstance were management measures deemed appropriate to achieve the desired outcome have been implemented, then instead of always questioning the adequacy of management, perhaps the first reaction should be to question the adequacy of the targets, particularly if the setting of these targets has been primarily influenced by our experience of the past. It may be that targets based on historical data are no longer appropriate under current marine environmental conditions, and it is the targets that may therefore need revision in the first instance, rather than the management measures.

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[3] The MSFD (EC, 2008) lists eleven qualitative "Descriptors" of Good Environmental status. Descriptor 1 relates to the maintenance of biological diversity, Descriptor 4 concerns the maintenance of food web structure and function and Descriptor 6 focuses on maintaining seafloor intregrity, which includes ensuring full functionality with benthic invertebrate communities.

The problems associated with any target revision are exacerbated by the increasing acceptance that human activities are causing global warming (Cook *et al.*, 2013). While it is probable that anthropogenic global warming (AGW) may have caused a persistent gradual warming of the Earth's oceans from as early as the 1920s, it is the marked increase in the rate of warming since the late 1970s that has generated most interest (Ting *et al.*, 2009). Sea temperatures in the Northeast Atlantic are now rising at an unprecedented rate and reaching unprecedented levels. Marine ecosystems within the region are moving into situations never-before-experienced (Meehl *et al.*, 2007; Cheung *et al.*, 2010), at least within the time frame for which marine observation data are available. Reassessing targets under these circumstances will inevitably involve extrapolation beyond the ranges of available data, which ever target setting approach is employed, and this will increase uncertainty over the resulting assessment outcomes.

If revision of targets could be perceived as a relaxation of ambition in terms of maintaining biological diversity, preserving seafloor integrity, or conserving the natural functioning of marine food webs, then the evidence to support such revision will need to be compelling. The purpose of this paper is to start collating such evidence. We review documented changes in the marine environment of the Northeast Atlantic, and the effects that these have had on different components of marine ecosystems in various regions across the area. We then consider the various indicators proposed for Descriptors 1 (biodiversity is maintained), 4 (food webs) and 6 (seafloor integrity) (EC, 2010), focusing on specific metrics proposed either by individual European Union Member States to fulfil these general indicator functions, or as "common indicators" proposed by OSPAR for assessing status at the scale of MSFD defined Regions (e.g. Northeast Atlantic) or Subregions (e.g. Greater North Sea, Celtics Seas). We assess how recent observed and future forecast changes in the marine environment might affect the targets currently proposed for these indicators.

Evidence of a changing marine environment

Changes in annual mean sea surface temperature throughout the last century have been recorded in different marine regions across the Northeast Atlantic. The Barents Sea was generally cooler during the late 19th and early part of the 20th century, warmer during the 1920s to 1950s, cooler again in the 1960s to 1980s and then warm again in recent decades (Drinkwater, 2006). Similarly, the Celtic Sea to Bay of Biscay region was relatively cool into the early 20th century, then from the 1920s to around the 1960s temperatures rose, before cooling once again between the early 1960s and 1980s, since when a sharp increase in temperature has been noted (Southward *et al.*, 2005; Philippart *et al.*, 2007a). The Nordic seas experienced a period of warmer conditions from the 1920s to the 1950s, followed by a cooler period during the 1960s, but since the 1970s, sea temperature has again increased (Toresen and Østvedt, 2000). In seas around Greenland and Iceland, sea surface warming has proceeded at a rate exceeding the global average rate of 0.2 °C decade^{-1} (Bindoff *et al.* 2007). With the exception of a brief warm period during the early to mid-1970s, cool conditions generally prevailed from the 1960s to the mid-1980s in the North Sea. Prior to this sea temperatures were warmer, and since 1985 a strong warming trend has been evident (Edwards *et al.*, 2002; Reid *et al.*, 2003). In the Baltic Sea centennial timescale analysis of reconstructions of climate proxies (annual maximum ice extent and the vertically and horizontally integrated annual water temperature) over the past 500 years indicate that the 17th and 18th centuries were the coldest while the 20th century was clearly the warmest having the least ice in winter (Hansson and Omstedt, 2008).

The close similarity between, and synchrony in, these changes occurring in different marine regions suggest common drivers operating at the spatial scale of the entire Northeast Atlantic. The Atlantic Multidecadal Oscillation (AMO), derived after de-trending long-term variation in mean SST, suggests a cyclical pattern in positive and negative temperature anomalies with a periodicity of approximately 60 - 80 y (Ting *et al.*, 2009; Edwards *et al.*, 2013). Because the AMO is derived by removing the positive trend in SST, thought to be associated with anthropogenic global warming, the AMO is considered to reflect residual natural variation in ocean temperature (Reid and Valdes, 2011). Over the period when instrumentation has been available, warm phases have been evident between 1860 – 1890, 1930 – 1962, and 1995 to the present day, while cool periods have existed between 1895 – 1925 and 1967 – 1990 (Edwards *et al.*, 2013). The AMO would seem therefore to underpin the main pattern of temperature change observed in the various different marine regions of the Northeast Atlantic described above. Whilst the period of instrumentation only reveals two full cycles of the AMO (Ting *et al.*, 2009), other sources of information, such as tree-ring reconstructions, ice-core records, and model simulations, suggest that such a low-frequency cycle in SST may have been occurring for at least 500y (Keenlyside *et al.*, 2008; Knight *et al.*, 2006; Fischer and Mieding, 2005), while theoretical modelling suggests that the AMO may actually have been in place for at least 8000y (Knudsen *et al.*, 2011). The origin of the AMO is not known, but it would appear to drive changes in climate across the Northeast Atlantic, rather than to be driven by climate itself (Enfield *et al.*, 2001; Knight *et al.*, 2006). Two important time scales require consideration in relation to Baltic Sea ocean-climate system dynamics: one associated with the water balance (salinity conditions), with an e-folding time of approximately 33 years, and the second linked to the heat balance (thermal conditions) with a time frame of approximately 1 year (Omstedt and Hansson, 2006).

Several anomalous events have also been documented within the North Atlantic, and the effects of these have spread across the disparate seas comprising the Northeast Atlantic Region. A cold-boreal event, associated with reduced SST and salinity and so referred to as the "1970s Great Salinity Anomaly" (GSA'70s), initiated north of Iceland in the late 1960s (Dickson *et al.*, 1988) by a freshwater/sea-ice pulse from the Arctic via the Fram Strait (Belkin *et al.*, 1998; Wadley and Bigg, 2006). The GSA'70s was noticeable for at least 14 y as it transited around the Atlantic sub-polar gyre (Dickson *et al.*, 1988), moving through the Northeast Atlantic during the late 1970s (Turrell *et al.*, 1992). The pulse of low salinity water reached western UK waters in 1975, transited into the northern North Sea in 1976, reaching the central North Sea in 1977 and the southern North Sea in 1979 (Edwards *et al.*, 2002). 1977 and 1979 were amongst the coldest winter SST temperatures recorded in the North Sea in 50y (Becker and Pauly, 1996). The GSA'70s was largely advective in its origins (Dickson *et al.*, 1988), although this view has been challenged by Wadley and Bigg (2006) who attribute the propagation of the GSA'70s simply to anomalous oceanic currents or surface fluxes. In the North Sea the GSA'70s was associated with reduced inflow of warm Atlantic water from the shelf-edge current, which would also have reduced nutrient influx (Turrell *et al.*, 1992). Two further similar low temperature, low salinity anomalies in the Atlantic have been documented, in the 1980s and 1990s

(Belkin, 2004; Wadley and Bigg 2006; Haak *et al.*, 2003) and at least one early event, around 1910, is suspected (Dickson *et al.*, 1988).

In the North Atlantic, the North Atlantic Oscillation (NAO) is the dominant atmospheric driver affecting marine environmental conditions, accounting for at least 33% of variance in sea level pressure (Dickson and Turrell, 2000; Sarafanov, 2009). The NAO index is calculated as the surface level difference in air pressure at two stations in Iceland and in the Azores (Hurrell and Desser 2006). Decadal trends in the NAO influence regional temperatures, precipitation and wind speed and direction over the North Atlantic (Hurrell, 1995). Changes in local hydro-meteorological forcing are thought to be strongly linked to the NAO (Dickson and Turrell, 2000; Reid and Planque, 2000). In the late 1980s, the North Sea experienced a warm-temperate event (WTE'80s), associated with increased SST and salinity (Edwards *et al.*, 2002) caused by increased inflow of warm Atlantic water (Heath *et al.*, 1991; Turrell *et al.*, 1996; Belkin *et al.*, 1998) and this coincided with highest recorded NAO index values recorded during the 20th century (Edwards *et al.*, 2002). In Irish waters, 23%, 16% and 9% of total variance in sea surface temperature can be attributed to variation in the AMO, East Atlantic Pattern, and the NAO respectively. The recent warm phase is at least 50% attributable to the current warm phase of the AMO, with most of the remaining cause attributable to anthropogenic global warming (Cannaby and Hüsrevoğlu, 2009).

Since the 1990s the North Sea has remained in a warm state and many authors consider that the WTE'80s may have initiated a 'regime shift' in the North Sea's ecosystem. Biological evidence supporting this hypothesis is introduced in the following sections, but if the WTE'80s has caused a 'regime shift' then it can be attributed to three main causes (Beaugrand 2004): change in local hydro-meteorological forcing (Beaugrand, 2003; Beaugrand and Reid, 2003; Reid *et al.*, 2003), displacement of oceanic biogeographic boundaries to west of the European continental shelf (Beaugrand, 2003), and increased oceanic inflow into the North Sea (Reid *et al.*, 2001a; 2001b; Reid *et al.*, 2003). These three processes are all inter-linked. Some authors suggest that at least three regime shifts have occurred in the North Sea, in 1979, 1988 and 1998, all triggered by varying environmental factors and coinciding with similar fundamental changes to marine ecosystems elsewhere in the north Atlantic and north Pacific (Weijerman *et al.*, 2005).

In the North Sea, regime shifts can be initiated both directly by atmospheric forcing and indirectly through changes occurring in North Atlantic, whereas in the Baltic Sea, regime shifts are primarily driven by atmospheric forcing (Dippner *et al.*, 2012). Such shifts can, in some areas, be stabilised by fisheries-induced feedback loops within the foodweb (Möllmann *et al.*, 2009). In the semi-enclosed Baltic Sea, large-scale atmospheric circulation not only directly affects the temperature regime, but also other hydrographic variables such as salinity and oxygen content, which for example, influence the reproductive success of species of marine origin living at the boundary of their distribution in the Baltic Sea (e.g. Ojaveer and Kalejs, 2005; Köster *et al.*, 2009). The evolution of salinity and oxygen regimes has been directly linked to atmospheric circulation: thus stronger than average westerly winds are linked to abnormally low salinity conditions and higher than normal oxygen concentrations in all areas in the Baltic Sea (Zorita and Laine, 2000).

Although the Earth has actually been experiencing a warming trend since 1880, since the 1970s the rate at which global temperatures have been rising has increased markedly. Concentrations of greenhouse gasses, such as carbon dioxide, in the Earth's atmosphere have been at unprecedented levels since the 1950s (http://climate.nasa.gov/evidence) and man's use of fossil fuels as an energy source is believed to be largely responsible for this increase. Since the mid-1990s, therefore, there has been increasing consensus that human activities have largely been responsible for the global warming experienced in recent decades (Bray, 2010; Cook *et al.*, 2013). The world's oceans have absorbed much of this heat (Levitus *et al.*, 2000; Levitus *et al.*, 2009).

Evidence of environmentally driven change in marine ecosystems

In this section we consider each major ecosystem component in turn and review changes that have occurred in these components that have been attributed to environment drivers as the cause of change.

Changes in the phytoplankton assemblage

Half the quantity of carbon dioxide assimilated each year in the biosphere is fixed by phytoplankton (Behrenfeld *et al.*, 2006). The warming trend in the atmosphere has affected the oceans, modifying surface temperatures, water currents and stratification (Revelle and Suess, 1957; Bindoff *et al.*, 2007; Belkin 2009). Theoretically, higher water temperatures should stimulate phytoplankton primary production. However, in oceanic waters rising sea temperatures will generate more wide spread and more persistent stratification, restricting the availability of nutrients (Sarmiento *et al.*, 2004). This could be offset to some extent in some coastal regions, where stronger thermal gradients between land and sea caused by the land responding more quickly to rising atmospheric temperatures than the sea, is likely to result in increased upwelling of deep nutrient-rich waters (Bakun, 1990). Increased riverine input of nutrients is also likely to contribute to rising phytoplankton productivity in coastal regions (Rabalais *et al.*, 2009). Overall though, the net result of warming seas is predicted to be a reduction in global phytoplankton production (Behrenfeld *et al.*, 2006), and there is some empirical evidence to support this (Morán *et al.*, 2010). However, spatial variability in phytoplankton biomass exceeds that of sea surface temperature, pointing to the presence of complex advection and mixing processes affecting nutrient supplies to the euphotic zone (Klein and Lapyere, 2009), and this limits our capacity to predict future changes (Chavez *et al.*, 2011).

Water temperature *per se* appears generally not to be a limiting factor for phytoplankton production (Fasham, 2002). However, the processes involved are affected indirectly by variation in sea surface water temperature and the response varies between colder and warmer parts of the region because of differences in the physical structure of the water column. In cooler seas, north of latitude 50°N, rising temperatures correlate with increased phytoplankton abundance whereas south of 50°N the opposite occurs (Reid *et al.*, 1998; Richardson and Schoeman, 2004; Behrenfeld *et al.*, 2006; Chavez *et al.*, 2011). Light levels and nutrient concentrations influence phytoplankton production in the euphotic zone of the Northeast Atlantic. Warmer water tends to stratify so that the euphotic zone becomes nutrient poor and rising sea surface temperature reinforces this stratification, further restricting any mixing that might introduce nutrients from deeper water layers. Cooler water to the north tends to be mixed so that the euphotic zone is relatively nutrient rich. Here warming generally raises rates of chemical processes that drive phytoplankton production, and might stimulate moderate stratification that can be beneficial to phytoplankton growth (Valdés *et al.* 2007).

Through the 20th century it has been suggested that surface chlorophyll concentrations may have declined globally (Boyce *et al.*, 2010), but this observation has not gone unchallenged (Mackas, 2011; Rykaczewski and Dunne, 2011; McQuatters-Gollop *et al.*, 2011). In more recent times, average oceanic chlorophyll concentration appears to have increased by 22% between the periods 1979-1986 and 1998-2002. The increase was most notable in inter-tropical regions during spring and summer, less marked at higher latitudes, while a decrease was observed in the oligotrophic gyres. The level of increase in the Atlantic Ocean was second only to that observed in the Indian Ocean (Antoine *et al.*, 2005). Subsequent studies also suggested a global increase of 4% from 1998 to 2003, with change greatest (10%) in coastal regions (Gregg *et al.*, 2005), associated with coastal upwelling (Bakun, 1990).

Across the North-East Atlantic, synchronous changes in phytoplankton abundance and composition have occurred in relation to large-scale hydrodynamic patterns (Goberville *et al.*, 2014). However, on a smaller scale, local specificities in phytoplankton dynamics and environmental influences have been found. In waters around both Greenland and Iceland, chlorophyll concentration, phytoplankton biomass and productivity have shown increasing trends (Li *et al.*, 2006; Sherman and Hempel, 2009). In Icelandic waters, particularly to the north and northwest of Iceland, phytoplankton biomass is greater, and primary production higher, in years when the inflow of warmer Atlantic water is stronger than in years when this influx is weaker (Gudmundsson 1998). In the Barents Sea warming sea temperature is also likely to stimulate increased primary production (Ellingsen *et al.*, 2008), but here phytoplankton biomass has increased in recent years, even though change in sea surface temperature has been minimal (Sherman and Hempel, 2009). In the Norwegian Sea, where recent rates of warming have been high, temporal trends in phytoplankton biomass and productivity have been inconclusive (Sherman and Hempel, 2009), but a positive relationship between chlorophyll concentration and stratification has been suggested (Reid and Valdes, 2011). In the Baltic Sea, warming was the key environmental factor explaining changes in phytoplankton communities observed between 1979 and 2011: the general increase in total phytoplankton biomass, Cyanophyceae, Prymnesiophyceae and Chrysophyceae, and decrease in Cryptophyceae (Suikkanen *et al.*, 2013). In another high-latitude region, the seas around the Faroe Islands, despite recent high rates of warming, little change in phytoplankton biomass and production has been detected (O'Brien *et al.*, 2008; Sherman and Hempel, 2009).

Despite rapid warming of surface waters in the Celtic Sea since the 1980s, little overall temporal trend in phytoplankton abundance and production was detected (O'Brien *et al.*, 2008; Sherman and Hempel, 2009). However, variability within the region was marked. Whilst defining no clear temporal trend, two periods of high phytoplankton productivity were identified in the early 1960s and 1990s linked to higher rates of influx of nitrate rich oceanic water associated with variation in the NAO (Heath and Beare, 2008). Variation in phytoplankton biomass and productivity in the coastal shelf seas of southwest Europe have primarily been linked to changes in the rate influx of nutrient-rich oceanic water associated with varying levels of upwelling (Valdes *et al.*, 2007; Pérez *et al.*, 2010), variation in the onshore advection of western ocean water linked to atmospheric forcing (Van Aken, 2001) and changes in stratification of oceanic and coastal waters associated with extreme cold winter weather (Somavilla *et al.*, 2009).

In the North Sea and in waters west of the UK, rising sea temperatures since the mid-1980s have stimulated an increase in phytoplankton biomass, particularly in winter, and extended the phytoplankton growth season, which has become characterised by a succession of short-lived blooms Reid *et al.,* 1998; McQuatters-Gollop *et al.,* 2007; Vantrepotte and Mélin, 2009). Since the late 1990s, however, rates of change in phytoplankton biomass and productivity may have declined (Sherman and Hempel, 2009), and different trends have been apparent in different parts of the region (Lindahl, 1995; Lindahl *et al.*, 1998; Cadée and Hegeman, 2002; O'Brien *et al.*, 2008). Indeed a recent study based on in-situ observations of chlorophyll concentrations and inorganic Suspended Particulate Material in distinct hydrodynamic regions of the North Sea has suggested a lack of trend in chlorophyll in most areas with the exception of the permanently mixed region in winter (increase) and an intermediate region in summer (decrease) (Capuzzo *et al.* 2015).

Marked changes in phenology have also been noted (Reid *et al.*, 1992; Reid *et al.*, 1998; Edwards *et al.*, 2001b; Edwards and Richardson, 2004; Edwards *et al.*, 2007). These changes have been linked to changes in sea temperature, wind intensity and direction, warmer winters, and associated shifts in oceanic biogeographic boundaries that have affected the influx of nutrients from more oceanic waters (Reid *et al.*, 2001a; Reid *et al.*, 2003a; Beaugrand, 2004; Carstensen *et al.*, 2005; Leterme, *et al.*, 2005; Weijerman *et al.*, 2005; McQuatters-Gollop *et al.*, 2007 McQuatters-Gollop *et al.*, 2009). For example, the inflow of nitrate-rich oceanic water into the North Sea also increased in the late 1980s into the early 1990s (Dahl and Danielsen, 1992) and this has been linked to a pulse in new primary production[4] that coincided with a perceived 'regime shift' within the pelagic food web (Heath and Beare, 2008). However other studies suggest that the increase in phytoplankton biomass at this time was largely independent of any change in nutrient concentration (McQuatters-Gollop *et al.*, 2007; Llope *et al.*, 2009). Conversely, passage of the pulse of oceanic water associated with the cold boreal event of the late 1970s, the GSA'70s, north through western UK waters then south through the North Sea was associated with particularly low phytoplankton biomass in Continuous Plankton Recorder (CPR) samples (Edwards *et al.*, 2002). Addtionally, an increase in the concentration of suspended sediments in the North Sea, linked to storminess and coastal erosion, over the last 25+ years has led to a decrease in water clarity, which reduces the light available for photosynthesis and changes energy fluxes through the marine food-web (Capuzzo *et al.* 2015). Climate change model scenarios suggest that winter storms may increase in future, which would likely result in an increase in sediment resuspension and bottom and coastal erosion leading to higher concentrations of inorganic suspended solids in the water column, a further decrease in water clarity and a decrease in phytoplankton production (Capuzzo *et al.* 2015).

Rising temperature-related decreases in phytoplankton biomass are associated with changes in phytoplankton composition, with the community becoming increasingly dominated by comparatively small pico-phytoplankton (Atkinson *et al.*, 2003; Daufresne

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[4] New primary production is production that incorporates nitrogen from nitrate, as opposed to utilising nitrogen from ammonia and other breakdown products of metabolism and decomposition. The former represents the first "use" of this nitrogen within biological components of marine food webs, whereas the latter represents recycling of nitrogen within the biological food web.

et al., 2009; Morán *et al.*, 2010; Finkel *et al.*, 2010). In the North Sea, warming sea temperatures since the 1960s have been associated with increased dinoflagellate and decreased diatom abundance (Leterme, *et al.*, 2005; Edwards *et al.*, 2006a), both positively and negatively correlated respectively with variation in the NAO (Edwards *et al.*, 2001a; Edwards *et al.*, 2006a). Hydrographic changes that have occurred in the North Sea since the late 1980s, and which continue to the present day, favour the growth and earlier succession of dinoflagellates (Edwards and Richardson 2004; Edwards *et al.*, 2006b), and increased dinoflagellate to diatom ratios have been noted in the southern North Sea (Hickel, 1998) and the English Channel (Widdicombe *et al.*, 2010; Hernández-Fariñas *et al.*, 2013). Despite this overall increase in dinoflagellate abundance, some of the more common *Ceratium* species have actually decreased in numbers in 2000 (Edwards *et al.*, 2009). Off the French Atlantic coast, interannual variation in phytoplankton diversity was related changes in the AMO (David *et al.*, 2012).

However, these studies, and the conclusions drawn, have not gone unchallenged. While Leterme *et al.* (2005) found a decline in diatoms and an increase in dinoflagellates in the Northeast Atlantic between 1958 and 2002, Hinder *et al.* (2012) found the opposite over a slightly longer time period, 1958-2009, an increase in diatoms and decline in dinoflagellates. Different species subsets were selected for analysis in the two studies: Hinder *et al.* (2012) examined twelve diatom and nine dinoflagellate taxa, and Leterme *et al.* (2005) examined six diatoms and five dinoflagellates. Only nine taxa were common to both studies; perhaps the apparently contradictory results regarding functional group dynamics was a consequence of differing taxa selection. Regardless of what exactly the changes that occurred were, both studies agree that changes in the abundance of different taxa took place.

As dinoflagellate abundance has increased, the frequency of harmful algal blooms (HABs) in the North Sea has also risen, with exceptionally high frequencies being recorded in the 1980s off Norway (Edwards *et al.*, 2006a); again, this has been linked to the NAO (Belgrano *et al.*, 1999). Increasing sea temperatures at higher latitudes appear to have facilitated the northward expansion of some warm-water phytoplankton species, and this could be encouraging more HABs in areas where previously these have been an infrequent occurrence (Dale and Nordberg, 1993; Thorsen and Dale, 1997).

In parts of the northern Baltic Sea, phytoplankton species composition during the spring bloom, particularly the relative proportions of diatoms and dinoflagellates has been linked to variation in NAO phase. However, in other areas within the Baltic Sea, the relationship between the proportion dinoflagellates in the phytoplankton community and the NAO index is weak lacking, or even contrary to the relationship observed in the north, suggesting that other factors are important in shaping the spring phytoplankton communities (Klais *et al.*, 2011). Analysis of a 30 year time series suggested that thick (>30 cm) and long-lasting ice cover favored diatom-dominated spring blooms, while mild winters with storms and thin ice cover (10 to 20 cm), supported blooms of the dinoflagellate *Biecheleria baltica* complex. However, because of the intricate interplay of local hydrodynamics and the dinoflagellate life cycle, the spatial extent of *B. baltica* complex blooms are constrained through dispersal limitation. Over recent decades, peaks of key phytoplankton groups have shifted about 10 d earlier in the northwestern Baltic Sea (Klais *et al.*, 2013).

Changes in zooplankton populations and communities

Zooplankton are ectothermic and tend to be adapted to specific thermal environments (Hirche *et al.*, 1997). The changing marine environment has therefore affected the abundance and composition of zooplankton, caused distributional shifts among dominant zooplankton species, modified zooplankton reproductive phenology and efficiency, and altered the structure of zooplankton communities (Reid *et al.*, 2001b; Beaugrand *et al.*, 2002; Beaugrand, 2004; Bunker and Hirst, 2004; Edwards and Richardson, 2004; Möllmann *et al.*, 2005; Hays *et al.*, 2005). For example, changes in the distribution of *Calanus finmarchicus*, associated with increasing sea temperature (Reygondeau and Beaugrand, 2011), have strongly affected overall zooplankton abundance and biomass in several regions of the North Atlantic (Planque and Fromentin, 1996; Pershing *et al.*, 2004). In recent decades, shifts in distribution by as much as 10° latitude have been noted among calanoid species with a more southerly, warmer water affinity (Beaugrand, 2005; Edwards *et al.*, 2006b; Beaugrand *et al.*, 2009). Some warm water zooplankton species have extended their range by as much as 1000km in just 40y (Beaugrand *et al.*, 2003). In some regions this has led to increases in zooplankton diversity, which have generally coincided in a reduction in zooplankton mean size (Beaugrand *et al.*, 2010). Examples of the consequences of warm-water species distributional shifts can be found in the increase in *Calanus helgolandicus* density in the North Sea and Bay of Biscay (Bonnet *et al.*, 2005; Helaouët and Beaugrand, 2007) and the appearance in the North Sea of the cladoceran, *Penilia avirostris*, normally found in Mediterranean waters (Johns *et al.*, 2005); the increase in *Centropages typicus* abundance related to warming temperatures in UK seas (Beaugrand *et al.*, 2007); the northward shift of *Temora stylifera* in to the Bay of Biscay (Villate *et al.*, 1997; Valdés *et al.*, 2007); and the northward shifts of the hyperiid amphipods *Themisto abyssorum* and *T. compressa* in the Barents Sea, linked to increased inflows of warm Atlantic water (Kraft *et al.*, 2010). In the English Channel, such shifts are believed to have led to the increase in species richness, linked to rising sea temperatures (Eloire *et al.*, 2010).

Over the period 1958 to 1996, the abundance of *Calanus finmarchicus* in the North Sea was negatively related to the winter NAO, but since 1996 this relationship appears to have broken down (Beaugrand, 2005). In recent decades, the decrease in *C. finmarchicus* has led to a 70% reduction in total zooplankton biomass between the 1960s and 1990s (Edwards *et al.*, 2006b; Edwards *et al.*, 2007). Since the late 1980s, the abundance of meroplankton (temporary planktonic larvae of benthic species) has increased and, through competitive interactions with the holozooplankton (permanent planktonic organisms), these might now have a strong influence on tropho-dynamics within the pelagic ecosystem (Kirby *et al.*, 2007). In the Norwegian Sea, zooplankton biomass has declined by 70% since 1995. Prior to 2002, variation in zooplankton biomass correlated closely with changes in the NAO, but since 2002, this relationship has broken down and zooplankton biomass has declined by over 50%; a reduction that was not anticipated given observed changes in the NAO (Melle, 2008). In the Nordic seas, zooplankton biomass north of Iceland is negatively influenced by the strength of the inflow of warm Atlantic water onto the northern shelf; in cold years when inflow is weaker, zooplankton biomass can be double the level recorded in warm years (Astthorsson and Gislason, 1995). This variation in local hydrography causes considerable variation in zooplankton community structure in Icelandic waters (Gislason *et al.*, 2009). In the Barents Sea, both field and modelling studies suggest a positive relationship between zooplankton productivity and sea temperature (Dalpadado *et al.*, 2003; Ellingsen, *et al.*, 2008). In the Baltic Sea, interannual variability in the abundance of small zooplankton species (*Eurytemora affinis* and *Acartia* spp.) is controlled by climate variability during winter (Dippner *et al.*, 2001), such that the abundance of the most dominant zooplankton species correlated either positively (large neritic copepods) or negatively (freshwater cladocerans) with salinity (Vuorinen *et al.*, 1998).

Species composition of the zooplankton community of the northern North Sea has undergone several marked changes. Over the period 1962 – 1970, and again between 1978 - 1983, sub-arctic species were more prevalent (Beaugrand 2004a), coinciding closely with the two periods of cooler sea temperatures described above. But changes in the community were rather more complex. The abundance of temperate pseudo-oceanic species declined sharply in 1962, at the start of the first cool period and remained in a semi-depressed state through to 1978 when the next period of cool conditions further reduced their abundance (Edwards *et al.*, 2001; Edwards *et al.*, 2002). The short warm period during the early 1970s reduced the abundance of subarctic species, but their populations recovered during the second cool period. However, since the mid-1980s, while sea surface temperatures have steadily risen, the abundance of subarctic zooplankton has progressively declined while the abundance of pseudo-oceanic species has increased (Beaugrand, 2004a) and the abundance of many warm water and oceanic zooplankton species has reached unprecedented levels (Lindley *et al.*, 1990; Greve *et al.*, 1994; Reid *et al.*, 1992). Calanoid copepod and neritic zooplankton diversity has generally been highest during the periods when sea temperature in the North Sea was warmer, and least during the cool periods (Beaugrand, 2005). Changes within the zooplankton community that have persisted throughout the recent period of higher than usual SST in the North Sea provide much of the evidence supporting the contention that the WTE'80s initiated a regime shift. Step-wise changes in several indicators of zooplankton community composition, diversity, structure and phenology occurred around the time of the WTE'80s (Reid *et al.*, 2001; Beaugrand and Reid, 2003; Beaugrand 2004; 2005) coinciding with major biogeographic shifts in nine Calanoid copepod assemblages (Beaugrand 2005). Changes in Calanoid copepod community structure were most obvious in the northern and central parts of the North Sea (Beaugrand 2004).

Oceanographic conditions in the central Baltic Sea are strongly influenced by atmospheric forcing. The unusual period of persistently strong westerlies since the late 1980s, which has caused an increase in average water temperatures and decreasing salinity, has brought about changes in the dominance of the mesozooplankton community, from larger (e.g. *Pseudocalanus acuspes*) to smaller copepods (e.g. *Temora longicornis* and *Acartia* spp.) (Möllmann *et al.*, 2005). The younger stages of *P. acuspes* are primarily affected by winter atmospheric conditions and water temperatures whereas older life-stages are more strongly influenced by deepwater salinity conditions and chnages in predation pressure (Otto *et al.*, 2014). Variation in the abundance dynamics of another other large copepod in the Baltic Sea, the arctic species *Limonocalanus macrurus*, has also been shown to be linked to climate variability with higher abundances recorded in the NE Baltic during severe winters and *vice versa* (Ojaveer *et al.*, 1998).

Pelagic cnidarians and ctenophores have increased in abundance globally in recent years (Mills, 2001) and jellyfish outbreaks appear more frequent (Purcell *et al.*, 2007). In some cases, an increase in the frequency of jellyfish outbreaks in recent decades in the Northeast Atlantic has been linked to hydro-climatic change (Lynam *et al.*, 2004; Attrill *et al.*, 2007). Increased sea temperature, rather than food availability seems to be one of the principal triggers (Purcell, 2005; Gibbons and Richardson, 2009), and this may explain why outbreaks of warm-temperate jellyfish species have become more common in the Northeast Atlantic in recent years (Licandro *et al.*, 2010) and in the Irish Sea (Lynam *et al.*, 2011).

Changes in benthic invertebrate populations and communities

Reviews of the response of benthic invertebrates to changing environmental conditions can be found in Reid and Valdes (2011). Here we summarise the general findings of these reviews and consider additional and more recent information. Harley *et al.* (2006) conceptualised the response of benthic organisms to climate change (Figure 1). Climate change directly affects the performance of individuals at various stages in their life cycle (components indicated in green in Figure 1) via changes in physiology, morphology and behaviour (components indicated in orange in Figure 1). At the population level, climate change can modify transport processes that influence dispersal and recruitment of larvae. Community-level effects (components indicated in blue in Figure 1) are mediated by interacting species (e.g. predators, competitors, etc.) and include climate-driven changes in both the abundance and the interaction strength of species. The ecological effects of climate change on benthos include shifts in the performance of individuals, the dynamics of populations and the structure of communities. These, in turn, can result in altered species distributions, biodiversity and productivity.

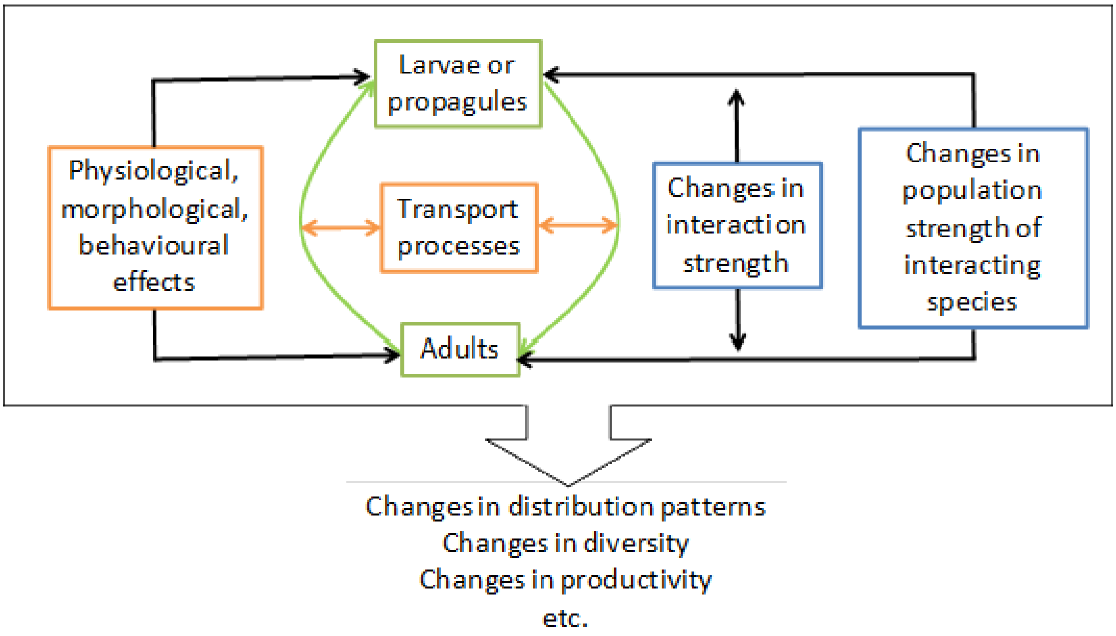

Figure 1. Potential ecological responses of benthic organisms to climate change (adapted from Harley *et al.*, 2006). Individual-level processes are shown in green, population-level process in orange and community-level processes in blue.

Benthic invertebrates are ectothermic; their physiology is therefore directly affected by variation in temperature. In order to remain within their optimal thermal niche, northward shifts in distribution in response to warming sea temperatures would be anticipated, while cooling temperatures might be expected to bring about the opposite change. Body size explains a large proportion of the variation in species' physiological responses to warming sea temperatures. However, idiosyncratic species responses, irrespective of body size, complicate population- and ecosystem-level predictions of the response to future climate change scenarios, particularly regarding the balance between variation in the proportions of large and small species within benthic communities, and variation in the proportions of large and small individuals within single species populations (Twomey *et al.*, 2012). Interpretation of observed changes in the distribution of benthic invertebrate species, and relating these directly to environmental stressors, is also complicated by the fact that benthic invertebrate communities are also strongly affected by human activities, especially fishing (Collie *et al.,* 2000, Kaiser *et al.*, 2006; Hiddink *et al.*, 2006, Tillin *et al.*, 2006; Reiss *et al.*, 2009). Despite this however, evidence of distribution shifts, or changes in abundance, in response to changing temperatures in the Northeast Atlantic can be found (Southward *et al.*, 2004; Eggleton *et al.*, 2007; Beukema *et al.*, 2009; Jones *et al.*, 2010; Witshire *et al.*, 2010).

Distributional responses have been particularly apparent amongst intertidal invertebrates perhaps because these organisms exist so close to their physiological tolerances. Populations of the Boreal barnacle *Semibalanus balanoides* around UK coasts increased in abundance during periods of cooling temperatures, but have decreased considerably during the recent warming period, while populations of two Lusitanian species *Chthamalus montagui* and *C. stellatus* have increased (Poloczanska *et al.*, 2008). In the Bay of Biscay, the southern range limit of *S. balanoides* has moved north (Wethey and Woodin, 2008), while the northern range limits of the two chthamalid species have moved north into Scottish waters (Mieszkowska *et al.*, 2006). Similarly, the southern barnacle species *Solidobalanus fallax*, not recorded in European waters until recently, has now expanded northwards to the English Channel (Southward, 1998; Southward *et al.*, 2004), while the range of another Lusitanian species *Balanus perforatus* has also extended northwards through the English Channel into the southern North Sea (Herbert *et al.*, 2003; Kerchhof 2002; Kerchhof *et al.*, 2009). Similar temperature-related distributional and local abundance changes among intertidal species include the gastropods *Osilinus lineatus*, *Gibbula umbilicalis* and *Testudinalis* spp. (Kendall, 1985; Kendall and Lewis, 1986; Mieszkowska *et al.*, 2006; Mieszkowska *et al.*, 2007), and the bivalve *Mytilus edulis* (Berge *et al.*, 2005). Among sub-tidal species, northwards shifts in distributional range in response to recent rising sea temperatures within the North Sea have been recorded for the Lusitanian decapods *Diogenes pugilator*, *Goneplax rhomboids* and *Liocarcinus vernalis* (Laporte *et al.*, 1985; d'Udekem d'Acoz 1991; 1997; Doeksen 2003; Franke and Gutow, 2004; Van Peursen, 2008; Neumann *et al.*, 2010), and the gastropod *Nassarius reticulates* (Craeymeersch and Rietveld, 2005). In the Barents Sea during the warm period of the 1920s to 1950s, the geographic ranges of several Arctic species contracted while the ranges of many Atlantic associated benthic invertebrates expanded northwards. In the more recent warming period blue mussels (*Mylelis edulis*) have appeared in Svalbaard after an absence of approximately 1000y (Berge *et al.*, 2005).

Weslawski *et al.* (2011) predict that with a changing climate, new patterns of species distribution will emerge among Arctic macrobenthos as boreal species invade more northerly waters and compete with functionally similar resident species. Since species will shift their distributions at their own intrinsic rates, the emerging patterns of competition will not just be between immigrant and resident species, but also between the immigrating species themselves. Species with long-lived planktonic larvae might be the first to arrive at new more northerly locations, but populations may not persist as their larvae are easily advected away from these new sites. Species with direct development may arrive more slowly, but once established are more likely to remain. While it is reasonable to predict that benthic assemblages will change as ocean temperatures rise (Kröncke *et al.*, 2011) and waters become more acidic (Table 8.1 in Reid and Valdés, 2011), the rate at which they will do so and the composition and function of the future assemblages is difficult to forecast. Benthic invertebrate species richness is higher in southern parts of the Northeast Atlantic than in the northern parts. These northward shifts of benthic species distributions might therefore bring about increased species richness in the more northerly parts of the region, such as in the North Sea. The assumption underlying this hypothesis is that more spcies will enter the North Sea from the south than will leave it to the north (Hawkins *et al.*, 2009; Beukama and Dekker in press).

In many instances, seabed habitats are critically defined by their resident biogenic fauna. Thus if change in the marine environment alters the population dynamics of such biogenic benthic invertebrate species, then profound changes in seabed habitat could result. In UK waters, for example, habitats dominated by predominantly northern species such as the horse mussel *Modiolus modiolus*, may decline and reduce their value as rich habitats for marine life. Others, characterised by southern species, for example the sea fan *Eunicella verrucosa* may increase in extent (Hiscock *et al.*, 2004). Scientific evidence on the effect of climate change on habitat-forming species is either sparse or inconclusive. Recently, a long-term CO2 perturbation study on the dominant reef-building cold-water coral *Lophelia pertusa*, thought to be susceptible to climate change, suggested that this species is capable of acclimatisation to acidified conditions (Form and Riebesell, 2012). Further ecophysiological studies of habitat-forming species are therefore necessary to ascertain whether the proposed targets for habitat-related indicators are feasible and attainable under prevailing climatic conditions (Gormley *et al.*, 2014).

Many benthic invertebrates pass through a pelagic planktonic (meroplankton) larval stage during the early part of their life-cycle. Apparently linked to rising sea temperature, the composition of benthic invertebrate meroplankton in the North Sea has changed in recent years; decapod and echinoderm larvae abundances have increased, while bivalve abundance has decreased (Kirby *et al.*, 2008), and timing of the seasonal peak has advanced by a month (Lindley *et al.*, 1993; Edwards and Richardson, 2004; Edwards *et al.*, 2009). Changes in temperature can directly affect gametogenesis and spawning of echinoderms (Kirby *et al.*, 2007), while lower metabolic rates in cooler winters can result in bivalves maintaining higher body mass and condition through to the spring, facilitating higher productivity rates leading to higher recruit densities (Beukema *et al.*, 1998; Strasser *et al.*, 2003). Conversely, rising sea temperatures reduce bivalve reproductive output and advance the timing of spawning (Honkoop and van der Meer, 1998; Philippart *et al.*, 2003).

Temperature related phenological advances in benthic invertebrate larvae abundance, if not linked to similar advances in their phytoplankton and zooplankton prey, could lead to a reduction in prey resources for benthic invertebrate larvae in the meroplankton (Òlafsson *et al.*, 1994): the match-mismatch hypothesis (MMH: Cushing 1990).

Planktotrophic benthic invertebrate larvae lack energy reserves and have high weight-specific metabolic demands. One study has reported clear effects of timing of spawning on the growth and development of benthic invertebrate larvae, but these were not related to variation in phytoplankton concentration (Bos *et al.*, 2006), while a second study demonstrated increased density dependent mortality among *Macoma balthica* larvae as the level of mismatch increased (Philippart *et al.*, 2003).

Benthic invertebrate communities are heavily influenced by prevailing hydrodynamic regimes (Butman 1987; Snelgrove and Butman, 1994; Wieking and Kronke, 2001; Kronke, 2006). Changes in ocean current can affect dispersal patterns during the meroplankton larval phase, to such an extent that dynamics of local populations can be significantly affected (Òlafsson *et al.*, 1994; Palmer *et al.*, 1996; Todd, 1998; Gaylord and Gaines, 2000; Svensson *et al.*, 2005; Levin, 2006). Variation in the physical and chemical properties of the water column can influence productivity, affecting the availability of food resources (Rosenberg, 1995). If warming seas result in a general decline in phytoplankton biomass and a reduction in size composition within phytoplankton communities (see above), then this is likely to result in a reduction in the export of fixed carbon to the seafloor as small phytoplankters are rapidly degraded in the water column (Bopp *et al.*, 2001). Even in the deep-sea ecosystems of the Northeast Atlantic, 4% of spring bloom surface production falls to the seafloor (Gooday, 2002) and variability in surface primary productivity is reflected in changes among benthic invertebrate communities (Davies *et al.*, 2007). In temperate stratified waters, primary and secondary production is elevated along thermohaline frontal regions where summer stratified waters are separated from permanently mixed waters (Hunt *et al.*, 1999: Scott *et al.*, 2006; Belkin *et al.*, 2009). The quality and quantity of organic matter falling to the seabed at frontal locations affects the growth, abundance, biomass and functional composition of benthic invertebrates (Rosenberg, 1995; Dauwe *et al.*, 1998; Amaro *et al.*, 2003; Amaro *et al.*, 2007). Any changes in sea temperature or water flows that affect frontal gradients or location could therefore have a marked effect on benthic invertebrate communities.

In the Baltic Sea, substantial structural change has occured within coastal zoobenthos communities since the 1960s and 1970s, such as declining abundance of marine polychaetes (e.g. *Bylgides sarsi*) and other species that are more sensitive to increased water temperatures (e.g. *Monoporeia affinis* and *Pontoporeia femorata*). These changes have been linked not only to large scale changes in climate, water temperature and salinity, but also to changes in the structure of local fish communities (Olsson *et al.*, 2013; Rousi *et al.*, 2013). Clear effects of changing water temperature on the abundance of several other species (e.g. *Saduria entomon*, *Potamopyrgus antipodarum*, *Mya arenari*a and *Hediste diversicolor*) have also been demonstrated (Rousi *et al.*, 2013), and there is evidence that changing weather conditions, such as wind speed and the number of calm days, has affected benthic invertebrate communities dominated by *Macoma balthica*, *Dreissena polymorpha* and *Amphibalanus improvisus* (Kotta *et al.*, 2009).

Oxygen depletion at or in the seabed is affected by both variation in temperature and water flow. Thermal stratification of the water column reduces water flow rates near the seabed. The extent of seafloor area that is hypoxic is expected to expand as sea temperatures increase, and this will markedly affect benthic invertebrate communities. Species diversity will decline as less tolerant species are lost and more tolerant opportunist species become increasingly dominant (Levin *et al.*, 2009). Mass mortality of benthic invertebrates has been reported following prolonged periods of hypoxia and it has been estimated that globally, 500,000 tonnes of benthic invertebrate biomass has already been lost because of recent expansions in hypoxic seafloor area (Diaz and Rosenberg, 2008; Seitz *et al.*, 2009). If changing climate results in a higher frequency of more intense storms (Alexandersson *et al.*, 2000; Ulbrich *et al.*, 2009; Donat *et al.*, 2010), then this will reduce thermal stratification of the water column and perhaps offset the potential detrimental effects of rising temperatures on seafloor hypoxia (Rabalais *et al.*, 2007). But, increased physical disturbance from wave stress could in turn pose a threat to some benthic invertebrate populations, particularly in shallow waters <50m in depth, such as in the southern North Sea (Rees *et al.*, 1977; Nehls and Thiel, 1993; Turner *et al.*, 1995; Posey *et al.*, 1996).

Over the last 200y, approximately 33% of anthropogenic carbon dioxide released into the atmosphere has been taken up by the oceans (Sabine *et al.*, 2004), making the oceans more acidic and reducing calcium carbonate saturation. During the 21st century increases in pH of 0.5 units over the pre-industrial level of 8.2 have been predicted (Caldeira and Wickett 2003; Blackford and Gilbert, 2007). Lower calcium carbonate saturation reduces calcification rates in marine organisms and the effects of this could permeate throughout entire marine ecosystems (Fabry *et al.*, 2008). The effects of ocean acidification on coral reefs has been reviewed (Langdon and Attkinson, 2005; Kleypas and Langdon, 2006). Cold water corals are found between depth of 200 and 1000m throughout the Northeast Atlantic (Reid and Valdes, 2011). They are closely adapted to a tight range in pH and can tolerate only very limited variation (Fabry *et al.*, 2008). They are perhaps the most vulnerable habitat forming calcifier in the region and major changes in abundance and distribution are likely to follow relatively minor changes in ocean acidity (Guinotte *et al.*, 2006; Turley *et al.*, 2007). Cold water coral reefs support abundant and diverse benthic invertebrate communities and changes in cold water coral calcification rates are expected to have major knock consequences for their associated benthic invertebrate fauna (Jensen and Frederickson 1992; Mortensen *et al.*, 1995; Husbø *et al.*, 2002). Globally, 70% of cold water corals are likely to be affected by the changes in ocean acidification predicted in the next 100y (Guinotte *et al.*, 2006). Palaeo-ecological studies suggest that in the past, such levels of change resulted in the extinction of a substantial proportion of benthic calcifiers (Zachos *et al.*, 2005).

Many other calcareous benthic invertebrates, such as echinoderms, bivalves, gastropods, barnacles, decapods, and foraminifera, are likely to have their capacity to develop their shells, endoskeletons and exoskeletons affected by increased ocean acidification (Harris *et al.*, 1999; Burnett *et al.*, 2002; Barry *et al.*, 2002; Gazeau *et al.*, 2007; Wood *et al.*, 2008; McDonald *et al.*, 2009; Lebrato *et al.* 2010). Species vary in their capacity to tolerate change in pH (Atkinson and Taylor, 1988; Batten and Bamber 1996; Shirayama and Thornton 2005; Widdicome and Needham 2007). Some species with high metabolic rate may be more disadvantaged because oxygen binding in the blood is pH sensitive (Pörtner and Reipschläger, 1996). Change in ocean pH is likely therefore to initiate considerable change in species composition within benthic invertebrate communities.

Changes in abundance, biomass, structure and function of benthic invertebrate communities have been linked to variation in the NAO (Frid *et al.*, 1996; Kröncke *et al.* 1998; Frid *et al.*, 1999; Kröncke *et al.* 2001; Weiking and Kröncke, 2001; Dippner and Kröncke, 2003; Franke and Gutow, 2004; Schröder, 2005; Rees *et al.*, 2006; Van Hoey *et al.*, 2007; Neumann *et al.*, 2008; Frid *et al.* 2009b). In the North Sea, several benthic species have suffered increased mortality, leading to marked reductions in abundance and consequent changes in functional and taxonomic community structure, and species diversity, following cold winters associated with a low NAO index (Ziegelmeier, 1964; Beukema, 1979; Kröncke *et al.*, 1998; Armonies *et al.*, 2001; Reiss *et al.*, 2006; Neumann *et al.*, 2009; Kröncke and Reiss, 2010). In the German Bight, cold winters initiated decadal scale reductions in abundance and diversity after severe winters (Schröder, 2005; Rehm and Rachor, 2007), whereas mild conditions linked to a positive NAO index resulted in increases in the abundance, biomass and species richness of benthic macrofauna (Beukema, 1990; 1992; Kröncke *et al.*, 2001). Similar correlations between the NAO and benthic invertebrate diversity and biomass have been noted in the western Baltic, Arctic fjords in Svalbard, and in the Kattegat and Skagerrak (Tunberg and Nelson, 1998; Beuchel *et al.*, 2006; Gröger and Rumohr, 2006).

Evidence of the 'regime shift', thought to have occurred within the North Sea phytoplankton and zooplankton communities, can also be seen in concurrent changes in the benthic invertebrate community (Reid and Edwards, 2001). Prior to 2000, benthic macrofaunal abundance, biomass, and species richness in the southern North Sea in spring were correlated with the NAO (Kröncke *et al.* 1998; Kröncke *et al.* 2001) and subsequent modelling studies suggested that atmospheric winter circulation over the North Atlantic was a strong predictor of macrofaunal community structure the following spring (Dippner and Kröncke, 2003). However, as with the phytoplankton and zooplankton communities, this correlation has subsequently broken down (Dippner at al., 2011). In the Kattegat in the Baltic Sea, which is both heavily exploited as well as being eutrophic, climate-driven changes in local environmental conditions (temperature and oxygen concentration) combined with a drastic reduction in nutrient concentration is thought to have driven a regime shift involving a change from pelagic to benthic regulatory pathways within the benthic invertebrate community. Variation in disolved oxygen concentration, which determine the survival of largely immobile benthic invertebrates, is thought to have been the primary driver for these changes, and this has been linked to changes in sea surface temperature (Lindegren *et al.*, 2012).

Changes in fish populations and communities

ICES has previously reviewed climate effects on fish stocks (Rijnsdorp *et al.*, 2010). Here we summarise some of the key observations from this earlier review and we also consider additional and more recent information. Fish are also ectothermic; their physiology is therefore also directly affected by variation in temperature (Brander 1994; 1995; Yoneda and Wright 2005; Graham and Harrod, 2009; Pörtner and Peck 2010). Temperature strongly affects reproductive processes, affecting the timing of maturation (Pawson *et al.*, 2000; Kjesbu *et al.*, 2010; Tobin and Wright, 2011), oocyte growth and development once mature (Korsegaard *et al.*, 1986; Olin and Von der Decken, 1989; Van der Kraak and Pankhurst, 1997; Pawson *et al.*, 2000; Yoneda and Wright; 2005), and the onset of spawning (Kjesbu *et al.*, 2010). Timing of migration can be sensitive to variation in temperature; for example in the Celtic Sea, flounder migration occurs up to two months earlier in cool conditions (Sims *et al.*, 2004). Temperature also influences digestion rates (Jones, 1974; 1978; Andersen, 2001). If fish ingestion rates are not limited by prey availability (Reilly *et al.*, 2014), but rather by their capacity to process food through the stomach (Bromley, 1991), then increases in water temperature offers opportunity for fish to raise their feeding rates. Consequently, most fish species prefer specific temperature ranges (Coutant, 1977; Scott, 1982) and tend to occupy marine regions defined by these. Globally, cod (*Gadus morhua*) occupy areas with a mean annual seabed temperature of 1°C to 11°C (Sundby, 2000). During warming periods, cod might therefore be expected to retreat from the southern part of their range (e.g. the North Sea) and to expand into cooler parts of their range (Blanchard *et al.*, 2005; Drinkwater, 2005; Rindorf and Lewy, 2006). Since thermal niches differ between different species, species responses to environmental change also vary (Pörtner and Farrell, 2008). Some elasmobranch species are very sensitive to changes in temperature (Brown, 2003), which can initiate complex behavioural responses (Sims, 2003). In the North Sea, declines in the abundance of southern affinity species has been associated with mass mortality caused by severe cold winters (Woodhead 1964a).

Distribution shifts are the most obvious response by fish populations to change in sea temperature (Brander *et al.*, 2003); warming trends cause northward shifts in range (Rose, 2005). Warming of the Northeast Atlantic in recent decades has coincided with northerly shifts in the distribution of many species generally associated with more southerly latitudes (Quéro *et al.* 1998; Perry *et al.*, 2005; Beare *et al.*, 2004), and these have been widely reported throughout the region. In the Barents Sea, during the warm period of the 1920s to 1950s the distributions of cod, haddock (*Melanogrammus aeglefinus*) and herring (*Clupea harengus*) all moved further north (Drinkwater, 2006) while in the more recent warming period, blue whiting (*Micromesistius poutassou*) have also moved further north (Berge *et al.*, 2005). In warm years the over-wintering area of capelin has extended further north and east than in cold years (Gjøsæter, 1998). In the Nordic seas, the distribution of Norwegian spring-spawning herring shifted northwards during the warming period of the 1920s, and southwards during the cooling period of the 1960s (Vilhjálmsson, 1997) and in the more recent warming period of the 1990s the capelin (*Mallotus villosus*) distribution has extended further northwest (Björnsson and Pálsson, 2004). In European western shelf waters, many recent expansions in distribution have involved non-commercial planktivorous species, for example boarfish (*Capros aper*) in the Celtic Sea (Pinnegar at al., 2002), Bay of Biscay (Farina *et al.*, 1997; Blanchard and Vandermeirsch 2005), and even around off-shelf seamounts (Fock *et al.*, 2002). In Iceland an industrial fishery for this species now operates.

In the North Sea, the centres of distribution of 36 demersal fish species of varied biogeographic affinity shifted north during a warming period between 1977 and 2001 (Perry *et al.*, 2005). Records of less common pelagic species in the North Sea, such as sardine (*Sardina pilchardus*) and anchovy (*Engraulis encrasicolus*) have increased in number markedly since 2000 as sea temperature has risen, and such records now come from as far north as the western Norwegian coast (Brander *et al.*, 2003; Beare *et al.*, 2004). The distribution of sole (*Solea solea*) in the North Sea has exhibited marked multidecadal fluctuations in their latitudinal centre of gravity due to combined effects of fishing and climatetemerature change (Engelhard *et al.* 2011). The North Sea is, however, a relatively enclosed coastal shelf sea; surrounded by land to the west, south and east, and by deeper water to the north, which might represent inappropriate habitat for many species being pushed north by rising sea temperatures. For some species, e.g. plaice (*Pleuronectes platessa*) and cuckoo ray (*Leucoraja naevus*), little latitudinal change in distribution has been noted, instead these species have tended to move into deeper, cooler water nearby (Perry *et al.*, 2005). In contrast, Engelhard *et al.* (2011) suggest that the movement of plaice into deeper water has resulted in a gradual latitudinal shift since the 1940s. Compared with the cooler period 1980 to 1989, cod and haddock occupied deeper parts of the North Sea between 1990 and 1999 when sea temperatures were higher (Hedger *et al.*, 2004), and a study of 28 North Sea demersal species, noted that the average depth of the whole assemblage increased at a rate of 3.6m per decade as sea temperatures warmed between 1980 and 2004 (Dulvy *et al.*, 2008). Changes in the distribution of pelagic species, such as herring, have also been noted (Corten and van de Kamp, 1992). Other species, such as whiting, have shown no change in range over the period since the 1920s (Kerby *et al.*, 2013)

In many instances, particular marine regions do not encompass the entire range of a species, or particular sampling programmes (groundfish surveys) only operate within a small part of species' ranges. Under these circumstances, shifts in distribution may manifest themselves as changes in species' population abundance. A northward shift in range appears as a reduction in population abundance in the south of the range and an increase in the north. Species at the northern edge of their range in Icelandic water increased in abundance during periods of warmer conditions (Toresen and Østvedt, 2000; Björnsson and Pálsson, 2004). Herring, near the south of their range in the Celtic Sea, declined in abundance during warming periods but increased during cooling, while sardine, at the north of their range, displayed opposite abundance trends (Southward, 1980; Alheit and Hagen, 1997). This switching appears to have been occurring since the 13th century (Southward *et al.*, 1988). However, during the cooling period of the 1960s, sardine abundance declined as expected, but the anticipated increase in herring abundance failed to materialise. At this time herring stocks were at an unprecedented low level, and this perhaps compromised recruitment potential and hindered recolonisation (Hawkins *et al.*, 2003). Following more recent warming, sardines are once again abundant and have spread into the Irish Sea. In the Bay of Biscay, the number of tropical species in the assemblage has increased in recent decades as sea temperatures have warmed (Quéro *et al.*, 1998), and sub-tropical species with wide distributional ranges have tended to increase in abundance, while populations of temperate species, or species with narrow distributional range, have tended to decline (Poulard and Blanchard, 2005). In the North Sea, two periods of increasing abundance of species with a more southerly affinity have been documented, in the mid-1970s and in the 1990s, both coinciding with positive water temperature anomalies (Corten and van de Kamp, 1996), although Beare *et al.* (2004) believe this to be part of a longer-term trend. Influxes of either new species, or species that have been absent from the North Sea for many years have also been reported (Pinnegar *et al.* 2008). These represent special cases of increased population abundance; increases from a start point of zero. Increases in the population size of species with a southerly affinity, which have regularly been recorded in the North Sea, such has horse mackerel, have also been documented (Reid *et al.*, 2001).

Fish species richness tends to decrease by approximately 1% with each increasing degree in latitude (Macpherson and Duarte, 1994). However the number of species showing northward range expansions with increasing temperature exceeds the number of species demonstrating range contractions, leading to an increase in species richness in more northerly latitudes. Demersal fish species richness in the North Sea has increased in recent years and this has been linked to rising sea temperatures associated with climate change (Hiddink and ter Hofstede, 2008; ter Hofstede *et al.*, 2010; Simpson *et al.*, 2011).

Many different mechanisms have been implicated as drivers of these local changes in abundance and distributional shifts. In the Nordic seas, changes in reproductive physiology appear to underpin the northerly displacement of cod spawning grounds at higher temperatures and their return south during cooler periods (Sundby and Nakken, 2008). Likewise, cod spawning is restricted to just the southern coast of Iceland during cool periods, but at higher temperatures, spawning occurs all around Iceland (Sæmundsson, 1934; Drinkwater, 2000). Larval survival also plays a role, cohort strength appearing stronger in warmer years and weaker in cooler years (Sundby and Nakken, 2008). Increased larval survival, combined with raised larval transport from Iceland in the Irminger Current (Jensen, 1949; Dickson and Brander, 1993), appeared to have driven the 1200 km northward shift in cod distribution off the Greenland coast during the 1920s to 1930s warming period (Jensen, 1939). Changes in larval distribution appear to be the predominant driver in the northerly shift of cod in the North Sea. Here temperatures experienced by adult cod have for some years been superoptimal in summer, but even though capable of moving considerable distances (Robichaud and Rose, 2004), the majority of fish do not move to seek cooler conditions (Neat and Righton, 2007). However, a series of warm winters with stronger than usual southerly winds during the egg and larval phase seems to have caused a northerly shift in the distribution of metamorphosing cod larvae, since the adults originating from these larvae have demonstrated a strong inclination to remain in these more northerly locations (Rindorf and Lewy, 2006).

Fish larvae appear to be particularly susceptible to variation in mortality related to changing environmental conditions (Letcher *et al.*, 1996). In the North Sea, larvae of clupeids, sandeels, dab and gadoids have been linked to changes in the plankton ecosystem, while larvae of migratory species (Atlantic mackerel) responded more clearly to hydrographic changes (Pitois *et al.*, 2012). Climate variability seems more likely to influence fish populations through bottom-up control via a cascading effect from changes in the North Atlantic Oscillation (NAO) impacting on the hydrodynamic features of the North Sea, in turn impacting on the plankton available as prey for fish larvae. Reduced flow rates in the Dooley Current in the early 1970s, linked to reduced inflow of Atlantic water through the Fair Isle – Shetland gap resulted in the failure of herring larvae spawned along the northeast coast of Scotland to reach their nursery ground in the south-eastern North Sea (Corten, 1986; 1990; Turrell, 1992), contributing to the collapse of the North Sea herring stock (Cushing, 1980). Variation in flatfish recruitment levels has been linked to changes in salinity in the southern North Sea (Beaugrand, 2004). The effect of changing environmental conditions on fish larvae survival rates may also be mediated through food web processes. Between 1963 and 1983 cod biomass in the North Sea was at almost unprecedented high levels (Cushing, 1982; 1984; Pope and Macer, 1996). This period coincided with a period of unusually high abundance of the zooplankton prey of cod larvae (Beaugrand, 2005), resulting in a prolonged period of higher than average recruitment to the cod stock (Greenstreet and Shephard, in prep). However, rising temperatures since the mid-1980s appear to have modified plankton communities, through changes in zooplankton abundance, size, and phenology, so as to reduce the survival rates of young cod (Beaugrand *et al.*, 2003). Over the long-term, changes in gadoid recruitment rates have paralleled variation in SST and plankton (Beaugrand, 2004). In the Baltic Sea between 1958 and 2012, in years when water temperatures were highest and the retreat of sea ice was earliest, peak herring larve abundance occurred up to five weeks earlier than in the years when ice retreat was later, but the timing of peak copepod *nauplii* remained unchanged, and this is thought to have affected herring recruit year-class strength through modification of temporal coupling between fish larvae abundance and the abundance of their prey (Arula *et al.*, 2014).

In the North Sea, increased water inflow during warming periods is thought to have contributed to increased larvae influx, causing increased abundance among populations of species with a southern affinity (Corten and van de Kamp 1996). Two 'invasions' have been particularly well documented: the bluemouth (*Helicolenus dactylopterus*) (Heessen *et al.*, 1996; Mamie *et al.*, 2007) and the snake pipefish (*Entelurus aequoreus*) (Lindley *et al.*, 2006; Harris *et al.*, 2007; Kloppmann and Ulleweit, 2007; van Damme and Couperus, 2008). Both bluemouth invasions, in 1991 and 1997, coincided with pulses of increased inflow of oceanic water into the northwestern North Sea, believed to have carried in eggs and larvae (Heath *et al.*, 1991; Edwards *et al.*, 1999). The snake pipefish outburst was associated with a marked increase in the occurrence of their eggs in the CPR as far west as the mid-Atlantic ridge, and this coincided with higher sea temperatures during the time when eggs and larvae were developing in the plankton (Kirby *et al.*, 2006). Increased food availability, arising from changes in the zooplankton, may have enhanced larval survival (van Damme and Couperus, 2008), contributing to higher larval immigration rates into the North Sea. Conversely, reduction in the inflow of Atlantic water into the North Sea in the late 1970s is thought to be at least partly responsible for the decline in herring abundance, the southerly shift in the sprat population and collapse of the fishery, and a reduction in the level of immigration of western mackerel (Corten, 1990; Corten and van de Kamp 1992; Svendsen at al., 1995).

Changing environmental conditions can also affect adult fish through alteration of food web interactions. When shifted to spawning grounds north of Iceland in response to warming conditions, cod condition tended to be higher because of a similar shift in the distribution of capelin, a principal prey of cod in the Nordic seas, resulting in increased overlap of the two species distributions and increased feeding opportunity for the cod (Saemundsson, 1934). Food webs with reduced overall primary production rates at their base, and where phytoplankton communities are increasingly dominated by smaller-sized organisms, are unlikely to be able to support the current fish biomass levels (Sherman and Hempel, 2009). Over the past 60y, there is some suggestion that catches of fish have become increasingly limited by primary production (Chassot *et al.*, 2010). There is also some suggestion that the spawning and recruitment of mid-trophic-level fish are most affected by increasing SST (Ottersen *et al.*, 2004).

In the western English Channel, nine of 33 core species in the assemblage displayed strong responses to changes in sea temperature (Genner *et al.*, 2004). However, increases and decreases in abundance of southern affinity species to warming and cooling temperature regimes respectively tended to be greater than the reverse responses shown by Boreal species, perhaps indicative of complex direct and indirect mechanisms in operation (Stenseth *et al.*, 2002; Genner *et al.*, 2004). Inter-annual variation in basking shark (*Cetorhinus maximus*) abundance off southwest Britain is generally positively correlated with NAO-associated variation on sea surface temperature (Cotton *et al.*, 2005). However, this correlation only holds at relatively large spatial scale (10-1000 km); at a more local scale variation in basking shark abundance is more closely linked to changes in the local abundance of *C. helgolandicus* (Sims and Merrett, 1997; Sims and Quayle, 1998; Sims, 1999; Sims *et al.*, 2000). It would appear that short-term and small-scale decisions are influenced by the immediate need to obtain sufficient prey, while longer-term, and larger scale choices are dictated by metabolic costs and the need to remain within an optimal thermal habitat (Crawshaw and O'Connor, 1997; Sims *et al.*, 2003).

Many of the changes in distribution and abundance described above have been linked to the transition between the cool AMO phase of the early 20th century to the warm AMO phase of the 1930s to 1960s (Drinkwater 2006). Along the west Greenland coast cod underwent a pole-ward range expansion of almost 1000km (Toresen and Ostvedt, 2000). The English Channel is close to the biogeographic boundary between herring and sardine; the warm temperate sardine is at the northern extent of its range while the cold boreal herring is close to the southern edge of its range. The warm AMO period favoured sardine, while herring stocks in the area almost collapsed (Southward *et al.*, 1988; Edwards *et al.*, 2013). Conversely, in the Nordic seas, where herring are close to the northern edge of their range, their biomass increased by a factor of 10 (Toresen and Ostvedt, 2000). When the AMO reverted back to a cooler phase in the 1970s, the herring population in the Norwegian Sea declined by four orders of magnitude, but the latest AMO warming phase has again seen an increase in herring biomass back to 1960s levels (Edwards *et al.*, 2013). The early 1930s AMO warming phase was also an exceptional period for the fish community of the North Sea, characterised by in influx of horse mackerel (*Trachurus trachurus*) from the Baltic, a marked increase in the anchovy stock and an increase in the abundance of southern affinity species similar to that observed during the most recent warming period (Reid *et al.*, 2001; Reid an Edwards, 2001; Beare *et al.*, 2004). The more recent AMO warming phase has also seen a three-fold increase in blue whiting (*Micromesistius poutassou*) biomass (Hátún *et al.*, 2009).

In some instances changes in fish population abundance, population distribution and community species richness and composition have been explicitly linked to the climatic processes that underpin some of the observed variation in sea temperature, salinity and water flow. For example, changes in the relative abundance of core species of the Bristol Channel fish community in the late 1980s were correlated with the NAO, while later changes among the occasional species in the community appeared more directly linked to changes in sea temperature, and possibly more closely related to the AMO (Henderson, 2007). Similarly, the Thames estuary is an important nursery ground for juvenile flatfish and assemblage structure, abundance of the dominant species, and fish growth rates have all varied in correlation with the NAO (Attrill and Power, 2002). Inter-annual variability in the Baltic Sea is strongly influenced by changes in the inflow of Atlantic water from the North Sea and is modulated by variation in the NAO. Weak inflows since the end of the 1980s have caused marine species to retreat from the northern and eastern parts of the region as a result of reduced prey abundance and a deterioration in the conditions that supported successful reproduction (Bagge *et al.*, 1994; Nissling *et al.*, 2002; Ojaveer and Kalejs, 2005). Changes in salinity and oxygenation were the principal factors affecting cod, while the effect of water temperature was less critical (Tomkiewicz *et al.*, 1998; Hjelm *et al.*, 2004; Köster *et al.*, 2005; Eero *et al.*, 2011). In contrast, changes in water temperature were more important in explaining variation in the abundance of annual recruit cohorts among sprats *Sprattus sprattus* and herring (Baumann *et al.*, 2006; Cardinale *et al.* 2009). Consequently, the shift towards greater dominance of Baltic Sea fish communities by species with a greater freshwater tendency, such as the pikeperch *Sander lucioperca* (Kjellmann *et al.*, 2001; Gröger *et al.*, 2007), since the 1970s has been linked to variation in the NAO index, as well as to changes in water salinity and temperature (Olsson *et al.*, 2012). Variation in water temperature has also been identified as a critical factor influencing the size and abundance dynamics of successive cohorts of perch *Perca fluviatilis* (Kjellmann *et al.*, 2001).

The temperature – size rule (TSR) is presented as a universal response to increasing temperature (Atkinson, 1994; Daufresne *et al.*, 2009) because of its well established physiological basis (Pörtner and Knust, 2007; Forster *et al.*, 2011) and general cross-taxa support (Gardner *et al.*, 2011; Forster *et al.*, 2012; Edeline *et al.*, 2013). Several previously established 'rules', such as Bergmann's Rule or James' Rule, which state that both between and within species, individuals of larger size are found in colder environments, can be interpreted as a result of temperature – size rule. Such temperature-size rules have primarily been applied to endotherms (Sand *et al.*, 1995; Freckleton *et al.*, 2003; Meiri *et al.*, 2003), but have also been observed in ectothermic taxa (Timofeev, 2001; Olalla-Tárraga *et al.*, 2006). Recent modelling simulations suggest incorporating this eco-physiological theory and projected trends in sea temperature associated with current climate change scenarios suggest that by 2050, average maximum body weight within fish communities will have declined by approximately 20% (Cheung *et al.*, 2013a). These conclusions have been challenged (Brander *et al.*, 2013) and subsequently refuted (Cheung *et al.*, 2013b), but most critically, the conclusions drawn from the original modelling approach have been supported by a recent empirical study. Over a 40 year period in which temperature in the North Sea has increased by over 1°C, concomitant reductions in asymptotic body size were observed in six of the eight species studied (Baudron *et al.*, 2014).

In considering the potential effects of environmental change on fish populations, a major difficulty lies in disentangling these from changes caused by anthropogenic drivers. For example, in the North Sea, fishing mortality has tended to be higher in the south than in the north (Heath *et al.*, 2003; Heath *et al.*, 2008) so northerly shifts in distribution could be expected to result from differential rates of depletion among spatially segregated sub-stocks (Hutchinson *et al.*, 2001; Wright *et al.*, 2006). However, examination of fish landings in the northeast Atlantic indicated that between the 1970s and 1980s more species displayed southerly shifts in landings than northerly shifts, while between the 1980s and 1990s the reverse was true. This coincided with the general warming of sea temperatures during the first period and the general cooling during the second period, tending to corroborate the over-riding influence of sea temperature (Heath, 2007).

Attempts to model the effects of climate change on fish populations have tended to focus on defining species' temperature tolerance, often based on the temperature 'envelope' of current distributional ranges (Pearson and Dawson, 2003). Climate change models are then used to forecast likely changes in atmospheric and sea temperature and predict the locations where these sea temperature conditions are likely to be found in the future. Such models suggest that local extinctions could well occur by 2050, especially in sub-polar regions and semi-enclosed seas. Pelagic species, such as herring and anchovy, might be expected to move pole-wards by up to 600km, while shifts in distribution of over 200km on average could be anticipated among some demersal species (Cheung *et al.*, 2009; Cheung *et al.*, 2011; Lenoir *et al.*, 2011). Projected warming trends in the Barents Sea, for example, is expected to cause northerly extensions in the range of many species, as well as driving a northerly extension in their spawning grounds (Drinkwater, 2005).

Changes in seabird populations and communities

Seabirds are endothermic and therefore less likely to be directly affected by changing atmospheric and sea temperatures, but a changing marine environment can still affect seabirds directly. Adverse physiological responses to rising temperature are not unknown and these can instigate behavioural responses that affect chick survival (Oswald *et al.*, 2008). Breeding ranges of species that perform long-distance foraging trips can be directly limited by temperature (Oswald *et al.*, 2011). Breeding success can also be directly affected by changes in the frequency of extreme weather events during the breeding period, while adverse conditions during the over-wintering period can also trigger acute high-mortality events (Aebischer, 1993; Frederiksen *et al.*, 2008; Mallory *et al.*, 2009). However, it is generally accepted that changing marine environmental conditions are more likely to affect seabirds indirectly, via bottom-up limitation processes mediated by changes cascading along food chains and affecting the availability and quality of their prey (Durant *et al.*, 2003; Wanless *et al.*, 2005; Frederiksen *et al.*, 2006; Kirby and Beaugrand, 2009; Luczak *et al.*, 2011; Luczak *et al.*, 2012; Burthe *et al.*, 2012).

Two rate processes principally affect seabird population dynamics, adult mortality and chick production; both are affected by environmental variation (Perrins and Birkhead, 1983). In long-lived species, such as many seabirds, variation in adult survival rate critically affects total life-time offspring production. In potential trade-off situations therefore, successful rearing of chicks in any one year may be sacrificed to safeguard the future survival of the adult birds (Lebreton and Clobert, 1991; Sæther and Bakke, 2000). This tends to make variation in annual chick production the more sensitive parameter to any change in environmental conditions that affect prey availability (Gill *et al.*, 2002). Kittiwakes (*Rissa tridactyla*) are amongst the most sensitive of seabirds to changes in food availability because of their small body-size, surface-feeding habit, restricted choice of prey, and tight energy/time budgets (Furness and Tasker, 2000); consequently many studies have focused on the effects of changing forage fish abundance on kittiwake breeding success.

Inter-annual variation in kittiwake breeding success in the Isle of May, southeast Scotland, correlates negatively with variation in sea surface temperature, not in the preceding spring period, but during spring of the previous year (Frederiksen *et al.*, 2004). Kittiwakes on the Isle of May primarily consume 1-group sandeels during April and May before switching to 0-group fish in June during the chick-rearing period (Harris and Wanless, 1997; Lewis *et al.*, 2001). The one year lag in the relationship between breeding success and SST has therefore been explained on the basis that changes in one-year old sandeel abundance or quality provide the intermediate vector, and these changes are imparted at around time the sandeels were metamorphosing as 0-group fish (Frederiksen *et al.*, 2004). Negative correlations between sandeel recruitment and both SST and sea bottom temperature have been demonstrated for both the North Sea as a whole and the western North Sea (Arnott and Ruxton, 2002). A statistical study of larvae sandeel data combined with recruitment data (in the Dogger Banks and Wadden Sea areas) suggested that while warm conditions may stimulate the production of sandeel larvae their natural mortality is typically greater and particularly so when there is abundant 1-year-old sandeel that are likely to be cannibalistic (Lynam *et al.*, 2013; Ritzau Eigaard *et al.*, 2014). Other environmental variables have also been linked to the breeding success of Isle of May kittiwakes. For example, the number of chicks raised per pair is positively related to the date at which the spring plankton bloom becomes established (Scott *et al.*, 2006) and the suitability of the marine environment to support *Calanus finmarchicus* (Frederiksen *et al.*, 2013). This suggests that the environmental influence on kittiwake chick productivity manifests itself right from the basal trophic levels of the food web. Sandeel recruitment in the North Sea is positively correlated with *Calanus finmarchicus* abundance (van Deurs *et al.*, 2009) and variation in sandeel larvae biomass off eastern Scotland has subsequently been shown to be positively related to local variation in diatom abundance and copepod biomass, and positively correlated with kittiwake breeding success in the following year (Frederiksen *et al.*, 2006; Frederiksen *et al.*, 2007).

Among 42 kittiwake colonies around the UK, distinct clusters of colonies were apparent within which chick productivity rates tended to vary synchronously. These clusters generally coincided with well documented sandeel spawning aggregations (Proctor *et al.* 1998; Pedersen *et al.* 1999; Munk *et al.* 2002), and kittiwake breeding success within each colony cluster was generally positively correlated with sandeel recruitment rates within each associated sandeel spawning aggregation (Frederiksen *et al.*, 2005). In colonies around Orkney, kittiwake breeding success was also shown to be negatively correlated with SST in the spring period of the preceding year (Frederiksen *et al.*, 2007). In the Barents Sea, kittiwake breeding success is positively linked to variation in capelin abundance and negatively correlated with the abundance of herring (Barrett, 2007). Recent increases in water temperature in the region has favoured an increase in herring abundance at the expense of capelin and this could be the distal cause of declines in northern Norwegian kittiwake populations reported in recent decades (Barrett, 2007).

Breeding success of other seabird species, such as puffins (*Fratercula artica*), razorbills (*Alca torda*) European shag (*Phalacrocorax aristotelis*), which also carry multiple prey items to their chicks, has also been positively linked to sandeel larvae biomass in the spring of the preceding year (so that it is the availability of these fish as 1-year old sandeels that is the key factor), while breeding success of guillemots (*Uria aalge*) has been linked to variation in the mean length of one-year old sandeels. The mean size of 0-group sandeels both in the sea around the Isle of May (Frederiksen *et al.*, 2006), and conveyed by adult puffins to their chicks on the Isle of May (Wanless *et al.*, 2004), has shown a long-term decline over more than 25 years, and this decline in the quality of fish prey brought to chicks has contributed to recent declines in seabird breeding success in a number of species, especially puffins (Wanless *et al.*, 2005). Long-term change in environmental conditions affecting both hatch date and early sandeel growth are believed to be the cause of this reduction in both the size of 0-group and 1-group sandeels in the sea off eastern Scotland (Wanless *et al.*, 2004; Heath *et al.*, 2012). In contrast to kittiwake breeding studies at the Isle of May, breeding success of puffins at a north-eastern Norwegian colony was positively related to variation in SST, because in years of higher SST, both the quantity and quality of their one-year old herring prey was increased (Durant *et al.*, 2003).

In four of five colonies situated to the west, north and east of mainland Britain, and in northern and north-eastern Norway, survival rates of adult Atlantic puffins (*Fratercula artica*) varied synchronously and was correlated with sea surface temperatures around each colony two years previously, and linked to the effect of sea temperature on recruitment of the major prey species consumed, herring (*Clupea harengus*), sandeels (*Ammodytes marinus*) and capelin (*Mallotus villosus*) (Harris *et al.*, 2005; Grosbois *et al.*, 2009). Puffin numbers at colonies in the north-western North Sea increased through the latter part of the 20th century, but since the early 2000s this trend has reversed (Harris *et al.*, 2009; Steel, 2009). Since 2000, puffins have increasingly used more northerly and westerly wintering areas, with a corresponding decrease in the use of southerly wintering areas. Conditions may have become less favourable in the North Sea in recent years, corresponding with the recent warming trend, and these changes appear to have been detrimental to the breeding populations involved (Harris *et al.*, 2010). However, use of these more distant wintering sites does not invariably incur increased mortality (Harris *et al.*, 2013). Synchrony in the survival rates of guillemots from colonies around the UK outside the breeding season was related to the over-wintering sites that they shared; suggesting differences in environmental conditions between different over-wintering areas had a strong effect on survival rate (Reynolds *et al.*, 2011). Overwintering survival among non-breeding juvenile guillemots has been negatively linked to variation in winter SST (Votier *et al.*, 2008).

Changes in the marine environment in recent decades have already had marked effects on the population abundance, mean size, distribution and phenology of *Calanus finmarchicus* (see section above). Modelling of future trends based on current climate change scenarios suggests that such changes will be ongoing, with the likely prognosis that *C. finmarchicus* will continue to decline across much of the northeastern Atlantic (Frederiksen *et al.*, 2013). The consequences for many seabird species, which depend on small mid-trophic level forage fish prey, for which *C. finmarchicus* is in turn a key prey species, are pessimistic. Many seabird species are unlikely to be able to maintain adequate chick productivity rates and are likely therefore to undergo population declines. As already intimated in this section, such effects may already be apparent in southern parts of the region, such as the northwestern North Sea, but this situation is likely to spread northwards, affecting most areas of the northeast Atlantic by the second half of the 21st century (Frederiksen *et al.*, 2013).

Even where environmental conditions might remain suitable for supporting *C. finmarchicus* other factors might have a negative impact on seabird breeding success. The distribution and local abundance of many fish species have also been affected by changes in the marine environment in recent decades (see section above), and this is also likely to have profound consequences of seabird populations. In some instances populations of fish species likely to compete with seabirds for their small mid-trophic level forage fish prey resource may increase. In other instances, populations of species such as herring may increase, and by preying on larval sandeels, this could have a negative impact on sandeel recruitment rates. Other fish species could also compete with sandeels for the *C. finmarchicus* prey resource, again potentially to the detriment of sandeel recruitment (Castonguay *et al.*, 2008; Prokopchuk , 2009; Langøy *et al.*, 2012; Frederiksen *et al.*, 2013).

Environmentally driven phenological changes have been recorded among many different species at various trophic levels in marine food webs (see sections above). Phenological change has been noted among seabird populations. In several species, for example, the date of the start of egg-laying has become increasingly delayed (Wanless *et al.*, 2008; Wanless *et al.*, 2009). This has been linked to changing environmental conditions (Votier *et al.*, 2009), and may well contribute to a miss-match between peaks in seabird energy requirements and the availability of their prey to meet these (Burthe *et al.*, 2012).

Variation in seabird survival rates have been linked to variation in the winter NAO and associated changes in SST for both kittiwakes and guillemots (Frederickson, *et al.*, 2004;

Lahoz-Monfort, *et al.*, 2011). However, such relationships have not been universally demonstrated. For example, variation in puffin breeding success, whilst linked to changes in SST and prey quantity and quality, was independent of variation in the winter NAO (Durant *et al.*, 2003). Links between major climate signals and variation in both adult survival rates and chick production rates have also been demonstrated in several other seabird species. Adult survival rates among northern fulmars (*Fulmarus glacialis*) has been negatively linked to variation in the winter NAO, while breeding success appears positively correlated with the NAO (Thompson and Ollason, 2001; Grosbois and Thompson, 2005). Numbers of lesser black-backed gulls (*Larus fuscus*) breeding along the Norwegian coast declined towards the end of the 20th century and into the beginning of the 21st century, and this decline, presumably associated with changes in adult survival rate, has been related to variation in the winter NAO, air temperature in winter, and the local abundance of 0-group herring (Bustnes *et al.*, 2010). In a study of a trans-equatorial migratory seabird, Cory's shearwater (*Calonectris diomedea*), at two colonies in the Mediterranean Sea, adult overwintering survival rates were correlated with the Southern Oscillation Index (SOI), while chick production rates were correlated with the NAO (Genovart *et al.*, 2013). A study of five North Atlantic seabird species documented a correlation between the NAO and adult survival. However, variation in SST explained a higher fraction of the variability in adult survival rate than variation in the NAO. Since the NAO effects were generally lagged, the study concluded that climate variability affected seabird survival only indirectly, through its effects initially on the marine environment, which then filtered through to seabirds via food web interactions (Sandvik *et al.*, 2005).

Numbers of breeding seabirds in UK colonies increased throughout most of the 20th century, but by the 1990s, this trend had diminished and many species were starting to show a decline (Mitchell *et al.*, 2004). This declining trend has continued into the 21st century with an average decline of approximately 7.5% between 1999 and 2010. Among ten species declines of over 10% have been recorded during this period (Daunt and Mitchell, 2013). As much of the discussion above implies, a changing climate, and the consequences of this for the marine environment and ecosystem, are generally believed to be primarily responsible for this reversal in population trajectories (Wanless and Harris, 2012). Modelling of seabird population dynamics based on current climate change scenarios strongly suggests that these negative trends may well continue (Frederiksen *et al.*, 2013). However, there is some evidence that the strength of bottom-up control of seabird population dynamics, and the effects of climate and marine environmental change on these population regulatory processes, varies between different marine regions (Lauria *et al.*, 2013). The consequences of a changing marine environment on seabird populations are unlikely therefore to be universal across the whole Northeast Atlantic.

Changes in marine mammal populations and communities

Like seabirds, marine mammals are endothermic and therefore likely to be better able to buffer the direct effects of changing water temperature. Given their capacity to thermoregulate and the presence of blubber as highly effective insulation, hypothermia is not anticipated to be a serious issue, particularly in the face of rising sea temperatures. However, the possibility of heat stress (hyperthermia) is thought to be a potential issue (Hokkanen, 1990).

Despite this though, a warming marine environment can still affect marine mammals directly. For example, in more northern areas of the northeast Atlantic, warming sea temperatures have had a marked impact on the southerly extent of sea ice cover, the duration of sea ice cover during the year, and actual quality of the ice. This decline in ice conditions since 2000 has become an iconic signal of climate change (Kovaks *et al.*, 2011). In the past decade annual sea ice reduction has occurred at three times the rate that was the norm over the previous three decades (Maslanik *et al.*, 2007) and in 2008, the extent of permanent ice cover was the lowest ever recorded representing a reduction of 40% (Nghiem *et al.*, 2007). The thickness of sea ice cover in the Arctic has declined by approximately 0.6m between 2004 and 2008 (Kwok and Rothrock, 2009). Expectations for the future are that summer sea ice will continue to decline, perhaps reaching an almost ice-free state by 2035 (Wang and Overland, 2009). These changes in sea ice conditions have already had direct effects marine mammals that rely on sea ice to haul out and reproduce, hide from predators or inclement weather within ice fields, or which prey on ice-associated fish and invertebrates (Tynan and Demaster, 1997; Barber and Iacozza, 2004; Ferguson *et al.*, 2005; Kovaks and Lydersen, 2008; Laidre *et al.*, 2008; Kovaks *et al.*, 2011).

Four species of seal, harp, hooded, ringed and bearded seals, all breed on sea ice in the late winter and early spring, and all four species are likely to be affected by changes in the extent, and quality of ice. In the Barents Sea, annual pup production of harp and hooded seals was estimated at approximately 360 000 pups between 1998 and 2003, but by 2009 and 2010 this had declined to around 160 000 pups per year, thought to be linked to a reduction in ice conditions in the breeding lairs from 2003 onwards (Haug and Øigård, 2012a). The population of hooded seals in the East Greenland Sea declined by over 80% between the 1940s and 1980 because of excessive harvesting (Haug and Øigård, 2012b) and from 1980 onwards exploitation levels were reduced until by 2008, all harvesting was prohibited. However, the population has continued to decline and reduced availability of suitable ice to support breeding is the most likely explanation (Evans and Bjørge, 2013). At the beginning of the twentieth century, the abundance of Baltic ringed seals (*Phoca hispida botnica*), inhabiting three distinct sub-basins in the northern Baltic Sea amounted to about 190 000 individuals, but by the late 1970s several factors such as intensive hunting and reduced fecundity linked to high organochlorine pollution levels had reduced the population to only around 5000 seals (Harding and Härkönen 1999). Ringed seal pup survival rates are related to variation in the area of sea ice. Warming winter sea temperatures are likely therefore to result in reduced sea ice cover and so hamper any potential for recovery in the size of the ringed seal population, with the southern-most sub-population in the Gulf of Riga likely to be the most severely affected (Sundqvist *et al.*, 2012).

Some cetacean species may also be directly affected by changes in sea ice conditions brought about by warming sea temperatures; species such as bowhead whale (*Balaena mysticetus*) beluga (*Delphinapterus leucas*) and narwhal (*Monodon monoceros*), for example, could be affected by changes in open water refugia within the ice fields (Tynan and Demaster, 1997; Harwood, 2001; Heide-Jørgensen and Laidre, 2004; Laidre and Heide-Jørgensen, 2005; Simmonds and Isaac, 2007). Rising sea levels linked to reduced ice cover could affect shallow water species such as tucuxi, humpback dolphin and finless porpoise, as well as species, such as the grey whale, that use shallow bays to give birth (IWC, 1997; Würsig *et al.*, 2002). Changes in sea ice may also not be the only habitat change linked to warming seas to affect seals. Rising water levels may, for example, affect the access of grey seals to cave breeding sites to the north and west of the UK, or restrict access to low lying coastal breeding and haul out sites in the North Sea (Harwood, 2001; Würsig *et al.*, 2002; Evans and Bjørge, 2013). Other direct effects of climate change on marine mammals could include changes in rainfall pattern, which affects the availability of pools in some grey seal breeding sites. Female grey seals aggregate around pools allowing a smaller number of males to monopolise the breeding female component of the populations. Variation in rainfall could therefore affect breeding dynamics with local populations, affecting local population genetics (Twiss *et al.*, 2007).

The ranges of most cetacean species are geographically limited (Rice, 1998). MacLeod (2009) suggests that the ecological niches of cetaceans are defined primarily by water temperature, water depth, and the distribution and abundance of their preferred prey. Of these three factors, the water temperature appears to be the primary determinant of geographic range (Kaschner *et al.*, 2006), thus for example, spotted (*Stenella attenuata*) and spinner dolphins (*S. longirostris*) are limited to warm tropical waters (Mann *et al.*, 2000), clymene dolphins (*S. clymene*) are restricted to tropical and sub-tropical and waters of the Atlantic (Perrin and Mead, 1995; Rice, 1998), white-beaked dolphins (*Lagenorhynchus albirostris*) are endemic to the cooler waters of the northern North Atlantic (Rice, 1998; Reeves *et al.* 1999), and bowhead whales (*Balaena mysticetus*) and narwhals (*Monodon monoceros*) are only found in cold arctic waters (Mann *et al.*, 2000). Water depth and prey availability dictate fine scale distribution, determining how individuals are distributed within their range (McLeod, 2009). The reason why cetacean ranges are geographically explicit and apparently related to water temperature is not understood and could be due to a direct relationship between water temperature and species-specific thermal limits, linked to the distribution of each species preferred prey, or determined by inter-specific competition between different cetacean species (MacLeod, 2009). However several authors have claimed, or assumed, that a mechanistic understanding of the relationship between water temperature and cetacean range is not necessary; it is only necessary to know that they are related. On this basis empirical models have been developed by defining the species' thermal limits based solely on observed empirical data; survey sightings of each species and water temperature in which these sightings occurred (MacLeod, 2009; Lambert *et al.*, 2011). These models suggest that northward range shifts in cetacean populations is a likely response to rising seawater temperatures (Simmonds and Elliot, 2009; MacLeod, 2009; Lambert *et al.*, 2011). MacLeod (2009) suggests that, given current climate change scenarios, the ranges of 88% of cetacean species are likely to change in response to the expected changes in the marine environment, and for 47% of species, these changes are anticipated to have unfavourable implications for their conservation. Species with a preference for non-tropical continental shelf waters are most likely to be at greatest risk.

Although only derived from empirical models, these predictions are supported by recent sightings and strandings data. Dolphin species normally associated with warmer waters, such as the short-beaked common dolphin and the striped dolphin, have extended their range further north along the shelf seas west of the UK and around into the northern North Sea (Evans *et al.*, 2003; MacLeod *et al.*, 2005). However, it is not clear whether this is a direct physiological response by these dolphins to the changing sea temperature regime, or whether it simply reflects the fact that the distributional ranges of the fish species that form the main prey of these dolphins, anchovy and sardines, have also shifted further north (ICES 2008, see text above). Several other cetacean species normally associated with warmer climes, such as Blainville's beaked whale, Fraser's dolphin, dwarf sperm whale, Cuvier's beaked whale and pygmy sperm whale, have also been recorded more frequently in UK waters in recent decades, and observed further to the north than would normally have been expected (Evans *et al.*, 2003; Dolman *et al.*, 2010; Deaville and Jepson, 2011). Examination of strandings data from northwest Scotland over the period 1948 to 2003 indicated that over the period 1965 to 1981, no new species per decade were recorded, while from 1988 onwards this increased to two new species per decade. These new species were generally associated with warmer waters while the species recorded prior to 1988 tended to occur more often in colder waters. Between 1992 and 2003 the relative frequency of white-beaked dolphin strandings, a species normally associated with cold water, declined, and this decline in the frequency of strandings matched a reduction in the frequency of survey sightings. In contrast, over the same period, standings of common dolphin, a warm-water species, increased (MacLeod *et al.*, 2005). In contrast to this trend, records in UK waters of cetacean species with distributions normally to the north of the British Isles have not increased in frequency in recent years (Evans *et al.*, 2003; Evans, 2008a).

One of the major consequences of these anticipated range shifts in response to warming sea temperatures is that the cetacean species diversity will be redistributed. At lower latitudes, species richness and diversity is expected to decline, while at latitudes greater than 40° North, richness and diversity is anticipated to increase (Whitehead *et al.*, 2008; Kaschner *et al.*, 2011)

Although the mechanism for these range shifts may not be precisely defined, environmentally driven changes in cetacean distribution and local abundance mediated through intervening cascading trophic interactions within the food web, of the sort described in the sections above, are well documented (Kenney *et al.*, 1996). The 1982-83 El Niño caused bottlenose dolphins (*Tursiops truncatus*) to expand their range from southern to Central California and movement of their prey, rather than the change in water temperature, was believed to have been the cause of this range expansion (Wells *et al.*, 1990; Wells and Scott, 2002). Bottlenose dolphins off northeast Scotland are at the northern limit of their distribution, and the range expansion into this region may also have been facilitated by improved local feeding conditions (Wilson *et al.*, 2004). Around the Faroe Islands, peaks in the harvest of long-finned pilot whales (*Globicephala melas*) coincide with periods of warmer sea temperatures, which is when local abundance of the pelagic squid *Todarodes sagittatus* and blue whiting (*Micromesistius poutassou*) is also greatest (Bjørge, 2002; Hátún and Gaard, 2010). Nor are such environmentally driven, food web mediated range expansions restricted just to cetaceans. An expansion in the range of Antarctic fur seals (*Artocephalus gazelle*) to recolonise Heard Island in the subantarctic Indian Ocean, following an increase in sea temperature and glacier recession, was also linked to an improvement in local food supply (Shaughnessy and Green, 1998). There has been some speculation that recent changes in the distribution of harbour porpoises in the North Sea may be related to a shortage of sandeels, a known prey item, and this could also account to a decline in body condition among porpoises stranded on Scottish North Sea coasts (MacLeod *et al.*, 2007a; MacLeod *et al.*, 2007b; Thompson *et al.*, 2007).

Food web mediated environmental effects on marine mammals are not limited to just distributional shifts and changes in local abundance. In waters off west Greenland, recent increases in sea temperature had cascading effects on sea ice coverage, residency of top predators, and on the abundance of important prey species such as Atlantic cod (*Gadus morhua*). Harbour porpoises in the area depend on locating high density patches of prey with high nutritive value. They have responded to the general warming in the area by increasing their annual residency period, consuming greater quantities of cod resulting in improved body condition and larger fat deposits in their blubber compared with an earlier colder period in the 1990s (Heide-Jørgensen, *et al.*, 2011). Social dynamics within cetacean population can also be affected by variation in prey abundance. Off northeast Scotland bottlenose dolphin pod size declined following reductions in the local abundance of salmon (*Salmo salar*) (Lusseau *et al.*, 2004).

Rising sea temperature is anticipated to affect marine mammal reproduction, particularly when mediated through changes in marine food webs affecting prey availability. Calf survival in North Atlantic right whales has been linked to climate-related variability in prey abundance (Green and Pershing, 2004). Body fat condition and fecundity are closely correlated with prey availability in female fin whales (Lockyer, 1986) and when prey are abundant, females give birth to calves in consecutive years, whereas when prey are scarce, the breeding cycle can be extended to as long as three years (Learmonth *et al.*, 2006). Ovulation may well be suppressed if a certain body condition can't be attained (Boyd *et al.*, 1999). Similarly, calving intervals in humpback whales has also been related to maternal body condition (Wiley and Clapham, 1993). Off the Galapagos Islands, rising sea temperatures are believed to reduce conception rates among sperm whales, with consequent declines in calve abundance, and this has been linked to reduced foraging success (Whitehead, 1997). Differences in reproductive success have been linked to feeding conditions in harbour porpoises (Read and Gaskin, 1990), Antarctic fur seals (Boyd, 1996; Würsig *et al.*, 2002) and in sirenians (Boyd *et al.*, 1999).

Increasing sea temperature has the potential to increase pathogen development, survival and transmission rates and of increasing the susceptibility of marine organisms to disease (Harvell *et al.*, 2002; Lafferty *et al.*, 2004) and this been implicated as possible a factor responsible for the worldwide increase in reports of disease affecting marine mammals (Geraci and Lounsbury, 2002; Burek *et al.*, 2008; Van Bressem *et al.*, 2009). The frequency and severity of toxic algal blooms, e.g. those producing domoic acid, is likely to increase as sea temperatures rise and increased rainfall and freshwater runoff leads to increased nutrient enrichment (Peperzak, 2003; Lafferty *et al.*, 2004; Van Dolah, 2007) and mass die-offs linked to fatal poisoning have been reported (Geraci *et al.*, 1999; Domingo *et al.*, 2002; Geraci and Lounsbury, 2002), for example in Mediterranean monk seals (Hernandez *et al.*, 1998). Evidence that recent demographic changes in the population abundance of the two seal species found in UK waters, an increase in grey seal abundance and a decrease in harbour seal abundance, is equivocal. Both changes could in some way be linked to climate-mediated changes within marine food webs that have influenced the availability of prey to each species, although other factors such as competition with fisheries for their food resources, recovery from epizootics may also be important (Evans and Bjørge, 2013). The recent observation of domoic acid, a neurotoxin derived from harmful diatomaceous algae, in faecal samples from harbour seals in the northwestern North Sea (SCOS 2011) could be interpreted as evidence of a climate-induced changes in the marine food web (see text in previous sections).

Peaks in occurrence of short-beaked common dolphins in the North Sea have occurred recently, in the 1980s and in the 1930s/1940s (Fraser, 1946; Bakker and Smeenk, 1987; Sheldrick 1989; Baines *et al.*, 2006; Camphuysen and Peet, 2006), all at times when the NAO was in a positive phase characterised by warmer seawater temperatures, milder winters with wetter and stormier conditions, and stronger westerly winds (Hulme *et al.*, 2002). Off eastern Canada and in the White Sea between Russia and Norway, harp seal (*Pagophilus groenlandicus*) pup mortality was negatively correlated with the NAO. Sea ice cover also correlated negatively with variation in the NAO index. NAO linked changes in sea ice cover were thought to have contributed to a decline in harp seal population in eastern Canada between 1950 and 1972, and then to their subsequent recovery between 1973 and 2000. Across the entire harp seal breeding region across the north Atlantic, satellite data suggest that sea ice cover has declined at the rate of 6% per decade from 1979 to 2011 (Johnston *et al.*, 2012). Variation in bottlenose dolphin group size off northeast Scotland has been linked to change in the NAO, mediated through variation in the availability of their prey.

Future global climate change scenarios

Global climate projections depend on emissions scenarios that incorporate predicted atmospheric concentrations of greenhouse gases, aerosols, and other pollutants. For the Fifth Assessment Report of IPCC, the Special Report of Emission Scenarios tested 40 scenarios with a range of emissions based on a variety of assumptions regarding driving forces such as patterns of economic and population growth, and technology development. Levels of future emissions are highly uncertain and it is unlikely that any single emissions path will occur exactly as described in the scenarios, therefore multiple scenarios should be considered in any analysis of future climate.

For climate modeling and the analysis of potential impacts, Representative Concentration Pathways (RCP) are being used. These are consistent sets of projections of only the components of radiative forcing (the change in the balance between incoming and outgoing radiation to the atmosphere caused primarily by changes in atmospheric composition). Using these RCPs, climate model simulations were carried out under the framework of the Coupled Model Intercomparison Project Phase 5 (CMIP5) of the World Climate Research Programme (IPCC5) in year 2100 relative to 1750. The RCPs include one mitigation scenario leading to a very low forcing level (RCP2.6), two stabilization scenarios (RCP4.5 and RCP6) and one scenario with very high greenhouse gas emissions (RCP8.5). So projections of future climate change are conditional on assumptions of climate forcing, affected by the shortcomings of climate models, and inevitably also subject to internal variability when considering specific periods. In effect, climate change projections do not include changes in natural drivers such as solar or volcanic forcing or natural emissions.

The IPCC report includes projections for the next few decades that demonstrate spatial patterns of climate change similar to those projected for the later 21st century but with smaller magnitude. Atmospheric CO2 concentrations are higher in 2100 relative to present day as a result of a further increase of cumulative emissions of CO2 to the atmosphere during the 21st century. For the oceans, changes in sea surface temperature, sea level, sea-ice extent and pH are expected. However, projected changes are substantially affected by the choice of emissions scenario (Figure 2). Also, the scale of the changes in climate will not be uniform, and regional and temporal variability in the observed and predicted effects are very likely.

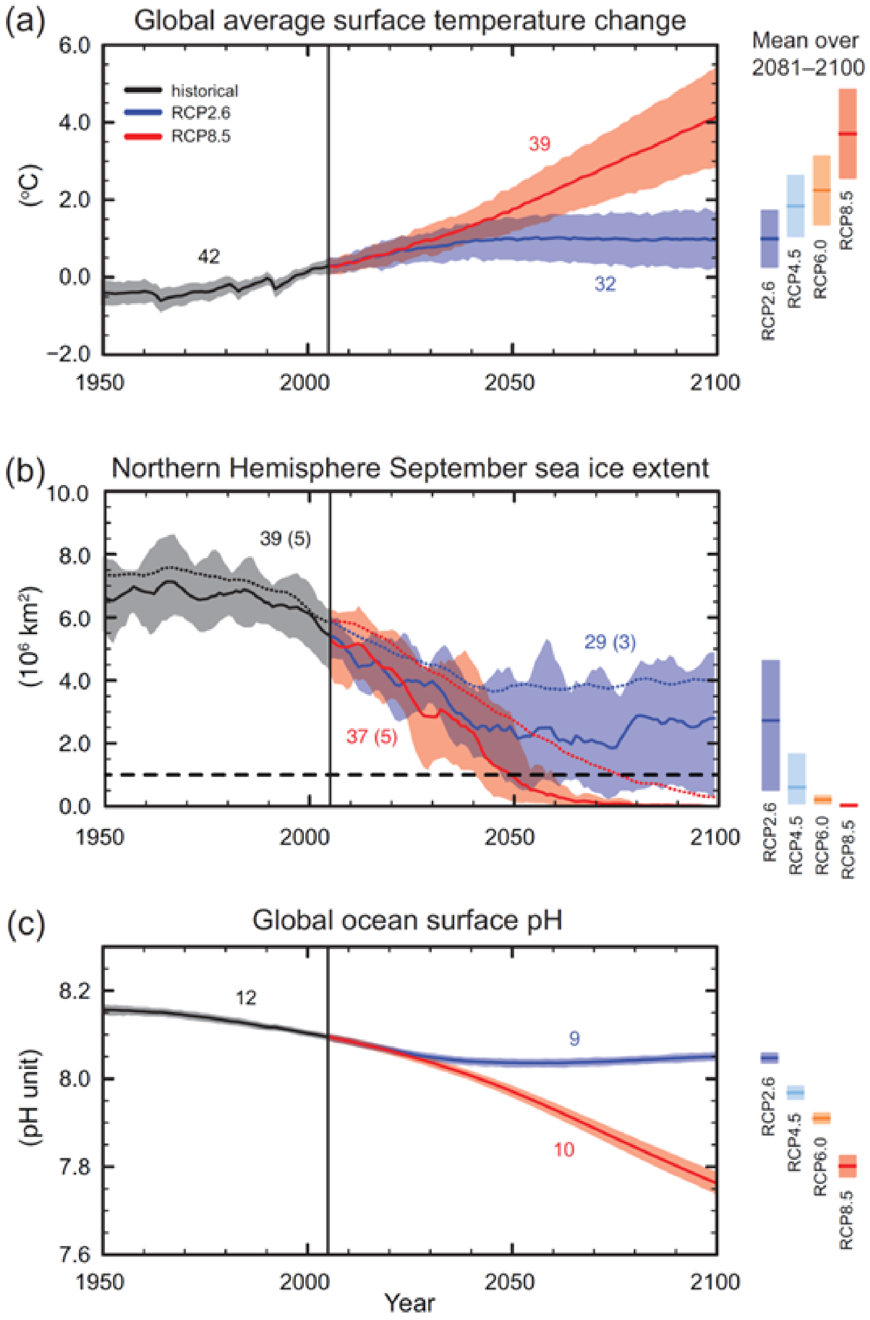

Figure 2. CMIP5 multi-model simulated time series from 1950 to 2100 for (a) change in global annual mean surface temperature (b) Northern Hemisphere sea ice extent (5-year running mean), (c) global mean ocean surface PH, and (d) global mean sea-level rise. Time series of projections and a measure of uncertainty (shading) are shown for scenarios RCP2.6 (blue) and RCP8.5 (red). The mean and associated uncertainties averaged over 2018–2100 are given for all RCP scenarios as coloured vertical bars. Source: IPCC 2013a.

Predicted changes in the marine environment

Global surface air temperature is going to increase with all scenarios. Even if the concentrations of all greenhouse gases and aerosols were kept constant at year 2000 levels, a further warming of about 0.1°C per decade would be expected. If emissions are kept within the range of the IPCC scenarios about twice as much warming (0.2°C per decade) can be expected. Best estimates of ocean warming in the top one hundred meters are about 0.6°C (RCP2.6) to 2.0°C (RCP8.5), and about 0.3°C (RCP2.6) to 0.6°C (RCP8.5) at a depth of about 1000 m by the end of the 21st century. The strongest ocean warming is projected for the surface in tropical and Northern Hemisphere subtropical regions. At greater depth the warming will be most pronounced in the Southern Ocean (high confidence). Compared to the predicted global increase of 1.6°C, annual temperature in offshore waters may warm in summer by between 2 and 4°C by the 2080s in the English Channel (Hulme *et al.* 2002).

It is very likely that the Arctic sea ice cover will continue to shrink and thin. In the Antarctic, however, a decrease in sea ice extent and volume is projected with low confidence for the end of the 21st century as global mean surface temperature rises. When using RCP8.5 (medium confidence; very high greenhouse gas emissions) scenario, a nearly ice-free Arctic Ocean in September before mid-century is likely (Figure 2). However, projections using other scenarios had very low confidence levels and thus resulted in wide ranging outcomes.

The geographical distribution of sea-level changes results from interactions between factors such as the geographical variation in thermal expansion, and changes in salinity, winds and ocean circulation. Consequently, as for many other variables, the range of regional variation in sea level rise is substantial compared with the global average. By the end of the 21st century, it is very likely that sea level will rise in more than about 95% of the ocean area with global mean sea level rise for 2081–2100 relative to 1986–2005 in the ranges of 0.26 to 0.98 m according to the RCP scenario. Sea level rise will not be uniform, however. About 70% of the coastlines worldwide are projected to experience sea level change within 20% of the global mean sea level change. From local simulations, for the U.K. for example, sea levels are predicted to be between 2 cm below and 58 cm above the current level in western Scotland and between 26 and 86 cm above the current level in southeast England by 2080, depending on the climate change scenario and effects of land movements.

The increasing uptake of CO2 is causing profound changes in seawater chemistry resulting from increased hydrogen ion concentration, referred to as ocean acidification (IPCC, 2013). In the IPCC assessment, Earth System Models projected a global increase in ocean acidification for all RCP scenarios. The corresponding decrease in surface ocean pH by the end of 21st century is in the range of 0.06 to 0.07 for RCP2.6, 0.14 to 0.15 for RCP4.5, 0.20 to 0.21 for RCP6.0, and 0.30 to 0.32 for RCP8.5. For the North Atlantic, Gehlen *et al.*, 2014 studied the potential magnitude of pH changes by 2100 and found that the spatial pattern in pH reductions is set by a combination of topography and North Atlantic circulation pathways. At the surface, the extent of ocean acidification was found to be set by the atmospheric CO2 trajectory, along with physical climate change, whereas, deep sea pH changes reflected atmospheric CO2 to a lesser extent. Whichever RCP model is used, over 17% of the seafloor area below 500m depth in the North Atlantic sector will experience pH reductions exceeding 0.2 by 2100. However, increased stratification in response to climate change could partially alleviate the impact of ocean acidification on deep benthic environments.

Climate change scenarios predict a weakening of the Gulf Stream during the twenty-first century, perhaps by as much as 25% by 2100, although a shutdown of the Gulf Stream is not predicted in any climate models (Hulme *et al.* 2002). Some preliminary studies on the potential change in surface currents in the north east Atlantic have been made for UK Climate Projections (Figure 3). The models suggest a potential reduction in the slope current by the end of the 21st century, but this conclusion needs to be treated with great caution due to the limited information available (Lowe *et al.*, 2009).

Other impacts of climate change in the marine environment include changes in salinity, storm frequency, wind speed and shifts in the locations of fronts and upwellings (e.g. Hulme *et al.* 2002, ICES 2004). In addition to global changes in salinity due to increased evaporation with increased temperature and changes in ocean circulation, additional localised modifications can result from changes in precipitation, associated river input and land run-off or the melting of ice sheets (Learmonth *et al.*, 2006). In European waters, trends in both increasing and decreasing salinity have been observed but whilst further changes are expected in the future, they are currently difficult to predict using climate models.

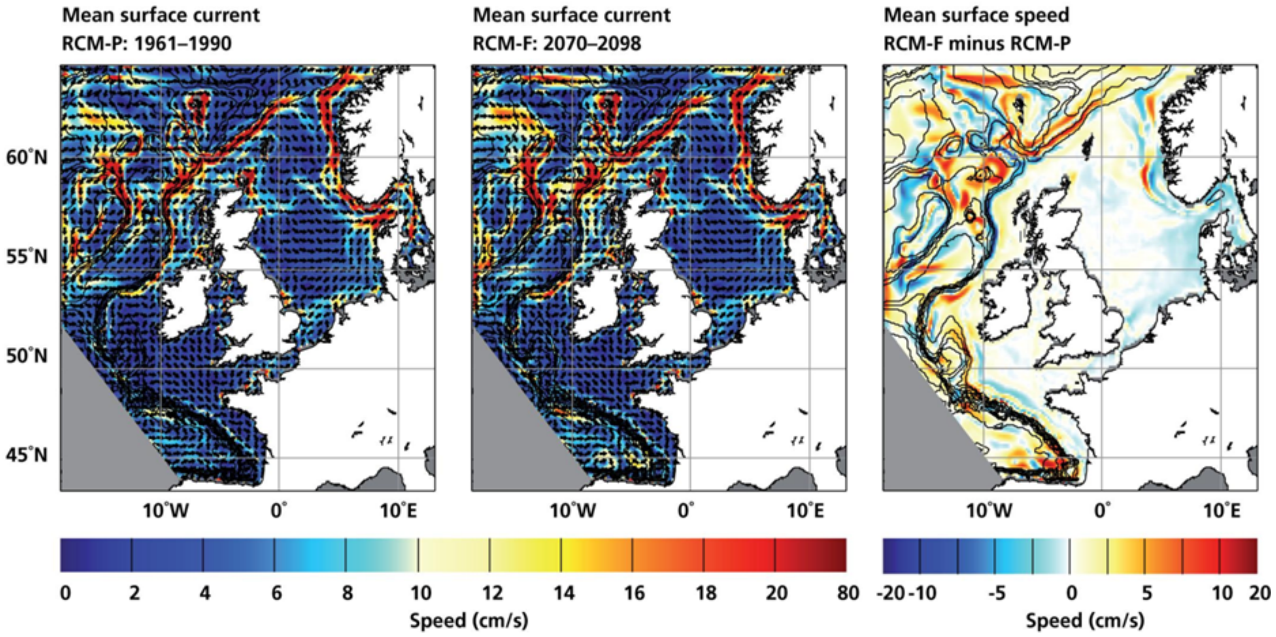

Figure 3: Mean surface currents from RCM-P (1961–1990), RCM-F (2070–2098) and the difference between them. Source: http://ukclimateprojections.metoffice.gov.uk/24066

Applications

The effects on the marine environment are especially difficult to predict because of the complex interactions between ocean processes and climate and consequently, the predictions of the effects on species and populations can be speculative (Würsig *et al.*, 2002). In any case, the application of scenarios of future climate change has enabled estimates of the potential redistribution of a species' climate space (Berry *et al.*, 2002; Pearson and Dawson, 2003). As an example, in the North Atlantic, current probabilities of occurrence and future geographic displacements of zooplankton (Reygondeau and Beaugrand, 2010), pelagic and demersal fish species (Beaugrand *et al.*, 2011; Lenoir *et al.*, 2011; Fernandes *et al.*, 2013) and benthic macroinvertebrates (Rombouts *et al.*, 2012) have been modeled and give an indication of the potential magnitude of the effects of climate changes for these groups. Changes in the biodiversity of exploited marine fish and invertebrates have been modeled in relation to changing sea temperatures (Cheung *et al.*, 2009). The projections demonstrate the potential for to numerous local extinctions in sub-polar regions, tropics and semi-enclosed seas, while simultaneously, species invasions occur most frequently in the Arctic Ocean and the Southern Ocean. Cheung *et al.* (2011) use the Special Report on Emission Scenarios (SRES) A1B scenario and incorporate the effects of changes in ocean biogeochemistry and phytoplankton community structure to project 120 species. The authors find an average rate of distribution-centroid shift of 52 km per decade northwards and 5.1 m per decade deeper from 2005 to 2050. Ocean acidification and reduction in oxygen content reduce growth performance, increase the rate of range shift, and lower the estimated catch potentials by 20–30% relative to simulations without considering these factors.

As projected changes in the environment are substantially affected by the choice of the emission scenario, so are the projections of future species distributions by climate model choice. Using scenarios from alternate climate models can yield contradictory results and this could lead to a misrepresentation of the vulnerability of species to climate change (Beaumont *et al.*, 2008). A possible method to select the appropriate emission scenarios from the wide range available is to apply a conservative and an extreme emissions scenario (Beaumont and Hughes, 2002; Araújo and Guisan, 2006). However, given that the more fossil intensive emission or "extreme" scenarios have already underestimated current emission growth (Rahmstorf *et al.*, 2007), the use of the relatively moderate to pessimistic A2 emission scenario is often justified (Rombouts *et al.*, 2012).

Implications for MSFD targets representing GES

In many instances, reference periods, or baselines established to determine the setting of targets for indicators are historical; indicator values prevailing at the time of this historical reference period are used as the basis for setting targets. The large fish indicator (LFI) in the North Sea provides a clear example of such a situation. Data to derive the LFI are available for the period 1983 to the present day. The early 1980s was the last period when ICES advice for fisheries management was to maintain *status quo*; i.e. to carry on fishing at levels prevailing at the time because these levels were deemed to be sustainable. On this basis the first data point in the LFI time series was adopted as the management target; an LFI value that would be consistent with GES in respect of North Sea demersal fish size composition (Greenstreet *et al.*, 2011). Development of the LFI in other marine regions has subsequently followed a similar logic (Shephard *et al.*, 2011; Modica *et al.*, 2014).

Similar examples can be found for seabirds and marine mammals. For example the OSPAR common indicators B1 “trends in relative abundance of breeding and non-breeding seabirds” and B6 “distribution pattern of breeding and non-breeding marine birds”, where it is proposed that baselines should be set as “a point in the past when, based on expert judgement, anthropogenic impacts are likely to be minimal compared to the rest of the time series”. In the absence of an appropriate historical reference point, long-term mean indicator values calculated over the whole time series might be used, but if the data show a defined trend, this would imply a ‘shifting baseline’. Even in these circumstances though, the target would be based entirely on historical data, but in this instance the whole data set. As a final option, where no historical data are available (i.e. a new monitoring programme), the first data point in the time series could be adopted as an interim target until a more robust approach can be determined (OSPAR, 2013).

Whatever method for target setting is employed, whether it relies on historic, recent or current indicator values, the resulting targets will still not take account of the effects of future environmental change on indicator values. It is entirely possible that such environmental change could affect the ecosystem component being assessed in a way as to make such targets unattainable. If some indicator assessments result in ‘failing GES’ outcomes, the first reaction should be to ensure that the target is appropriate, and adequately reflects prevailing environmental conditions, rather than simply assuming that management has been inadequate and immediately introducing even more stringent measures. In this section we start the process of compiling the documentary evidence that will be required should a case for altering targets need to be made.

We have reviewed a subset of the indicators proposed by regional seas conventions and the following case studies refer only to the OSPAR common indicator set. However, numerous indicators have been proposed by HELCOM (HELCOM 2013) and targets for these indicators will similarly require evaluation in light of climate change, particularly given the strong gradients in salinity and temperature present in the Baltic Sea. The majority of the HELCOM core biodiversity indicators differ from the OSPAR common indicators. However, there are some indicators that are the similar between the regional seas (e.g.: the HELCOM “Proportion of large fish in the community” and OSPAR “LFI”; HELCOM “Abundance of fish key functional groups” and OSPAR “fish dietary guilds”) and the case studies discussed below might be considered relevant for both areas.

HELCOM (2013) identify the following biodiversity indicators as linked moderately to pressure from temperature and salinity: Abundance of key fish species; Abundance of fish key functional groups; Zooplankton mean size and total abundance; Population structure of long-lived macrozoobenthic species; Cumulative impact on benthic habitats; Extent, distribution and condition of benthic biotopes; State of the soft-bottom macrofauna communities (salinity only); Lower depth distribution limit of macrophytes species (temperature only).

Pelagic Habitats

Three indicators are under development for pelagic habitats in the Northeast Atlantic MSFD region. The indicators each represent a different aspect of plankton community change and are meant to be used as a suite to inform Descriptors 1, 4 and 6. The pelagic habitat indicator development approach addresses pelagic habitat change at descriptor level rather than criteria level. In other words, multiple indicators inform each descriptor and some criteria. Additionally, because of the fundamental role of plankton in the pelagic ecosystem, the proposed indicators may also be used to inform and support other descriptors, for example, D2, D3 and D5. As of early 2015, all three pelagic indicators are still in development. All three proposed indicators are likely to be affected by climate change in a similar fashion, so we first present the individual indicators, and then in a summary section, we discuss the targets in the context of climate change.

Plankton lifeforms (PH-1)

This OSPAR indicator links to indicators 1.4.1, 1.4.2, 1.6.1, 1.7.1, 4.3.1, and potentially to 6.2.2 (EC, 2010).

PH-1 is comprised of time-series of pairs of plankton functional groups, or lifeforms. Change in the ratio of pairs can be examined over time. Although not identical, PH-1's approach has similarities with the UK's approach (Gowen *et al.* 2011). Plankton functional groups are often favoured as indicators over single species since indices of species abundance are frequently subject to large inter-annual variation, often due to natural physical dynamics rather than anthropogenic stressors (de Jonge, 2007). Functional group abundance is often less variable because variability in the abundances of the group's constituent species averages out. Moreover, indicators based on functional groups have been proved relevant for the description of community structure and biodiversity and more easily comparable than species-based indicators (Estrada *et al.* 2004; Gallego *et al.* 2012; Mouillot *et al.* 2006; Garmendia *et al.* 2012).

A variety of plankton lifeform pairs are undergoing testing for suitability as indicators. The proposed lifeform pairs have ecological relevance, as explained in Table 1. PH1 is also a foodwebs indicator (FW5).

Table 1. Potential lifeform pairs undergoing testing for suitability as indicators for "Common Indicator" PH-1. Shifts in the relative abundance of the two components of each lifeform pair are indicative of ecological change.

Life form pair	**Ecological rationale**
Diatoms and dinoflagellates	Dominance by dinoflagellates may be an indicator of eutrophication and result in less desirable food webs.
Gelatinous zooplankton and fish larvae	Indicator of energy flow and possible trophic pathways
Holoplankton and meroplankton	Indicator of strength of benthic-pelagic coupling
Large (<20um) and small (<20 um) phytoplankton	Size-based indicator of the efficiency of energy flow to higher trophic levels
Diatoms and non-heterotrophic dinoflagellates	Shift in primary producers may indicate eutrophication.
Harmful algal bloom (HAB) dinoflagellates and non-HAB dinoflagellates	Shift in phytoplankton community towards/away from dinoflagellate HAB taxa
Non-carnivorous zooplankton and phytoplankton	Indicator of energy flow and balance between primary producers and primary consumers
Non-carnivorous zooplankton and zooplankton carnivores	Indicator of energy flow and balance between primary consumers and secondary consumers

Pelagic diatoms and benthic diatoms	Indicator of benthic disturbance and frequency of resuspension events
Gelatinous zooplankton and crustaceans	Indicator of energy flow and possible trophic pathways
Large (> 2.0mm)copepods and small (<1.9mm)copepods	Size based indicator of food web structure and energy flows

Plankton biomass and/or abundance (PH-2)

This OSPAR indicator links to indicators 1.6.2, and 4.3.1 (EC, 2010).

Pelagic phytoplankton are responsible for the majority of marine primary productivity, making phytoplankton biomass a commonly-used indicator for primary production. Zooplankton are both consumers of phytoplankton as well as prey for higher trophic levels. Plankton biomass and plankton abundance are measured in different ways both within and between member states; however, as with PH-1, relative changes (trends) in time-series can be compared between datasets. In the Northeast Atlantic, phytoplankton biomass is estimated through phytoplankton carbon from biovolume, particulate organic carbon, chlorophyll concentration or the Continuous Plankton Recorder's Phytoplankton Colour Index (PCI); it is foreseen that all of these metrics will be used to inform PH-2. Zooplankton biomass is generally estimated by the carbon content of zooplankton samples; however the Continuous Plankton Recorder doesn't record zooplankton biomass outright, although it can be derived from CPR zooplankton abundance counts. Copepod crustaceans are a key zooplankton group which is trophically important, abundant, nearly-ubiquitous, well-studied, consistently monitored and taxonomically well-defined. For these reasons total copepod abundance will be the metric used to inform PH-2.

Plankton biodiversity indices (PH-3)

This OSPAR indicator links to indicators 1.6.1, and 1.7.1 (EC, 2010).

Biodiversity indices allow an accurate description of pelagic assemblages and also the direct comparison of communities that have few or no species in common. Diversity indices can describe the impacts of water pollution on biotic communities, which often affect only the structure of the assemblages or the abundance of a single or few species, and not the biomass or the ratio between functional or size groups. However biodiversity indices require standard observation methods based on microscopy and a high level of expertise in taxonomic identification (which is very time consuming), because the sensitivity values are assigned at species level (Lugoli *et al.* 2012; Garmendia *et al.* 2012; Estrada *et al.* 2004; Gallego *et al.* 2012). Diversity indices which focus on the number of species (or richness) (Menhinick Index) and dominance within the community (Hulburt Index; H) have been selected for testing. Currently, data from classic analytical techniques (microscopic counts) are being used to inform the indices. These allow only a fraction of the community to be quantified, underestimating true diversity. As plankton time-series expand in breadth to identify and enumerate microplankton, picoplankton, nanoplankton and bacteria, indices will increase in completeness.

Meeting pelagic habitat targets against a background of climate change

The three pelagic indicators share the same target "The plankton community is not significantly influenced by anthropogenic drivers". The target was constructed to allow for climate-driven changes in the plankton, yet trigger management action if a change in an indicator is linked to anthropogenic pressure. In other words, the changes in climate will not prevent the achievement of GES as the target was explicitly constructed to only fail if the plankton community responds to anthropogenic pressure. The first challenge in assessing GES is to determine whether an observed change in an indicator is caused by anthropogenic pressure. In order to separate anthropogenically-driven changes in the plankton from those caused by climate drivers, a spatially comparative approach is under development. Large scale changes in the plankton are most likely the result of large scale drivers, such as climate, while local scale change in the plankton may be the result of a direct anthropogenic driver, such as nutrient loading (McQuatters-Gollop and Vermaat, 2011). Therefore spatial comparison between indicator time-series may reveal regions exhibiting anomalous patterns which could be responding to anthropogenic, rather than regional climate, pressures. The second challenge for assessing GES for pelagic habitats is that a variety of plankton datasets employing differing sampling and analysis methods exist in the OSPAR region (including both fixed point monitoring stations as well as Continuous Plankton Recorder routes). This means that time-series from different member states, and even within some member states, are not directly comparable. In order to assess the state of the pelagic habitats across these disparate plankton datasets a trend-based, rather than numerically-quantitative, approach is under development. To address these two challenges, the Good Environmental Status assessment method will therefore compare relative patterns in plankton indicator time-series between spatial areas.

Benthic Habitats

Benthic habitats and communities vary annually, seasonally and spatially; detecting anthropogenic effects can be problematic because these can be masked by more dominant natural drivers (Løkkeborg, 2005). Most benthic habitat indicators have been designed and used to differentiate anthropogenic impacted sites from undisturbed reference sites. Consequently, the effects of a variety of anthropogenic pressures on the performance of indicators have been tested and described extensively, whereas information on their natural variability is scarce. Kröncke and Reiss (2010) demonstrated that knowledge of the natural dynamics of benthic indicators is essential for the assessment and monitoring of environmental status and for defining reference and baseline conditions. Long-term data on the natural dynamics of benthic indicators are particularly insightful when the management objective is to re-establish the historical structure of benthic habitats and their communities.

Typical species composition (BH-1)

Two different targets are covered under this indicator, which include:

i. its implementation as a state condition indicator by using an unweighted list of typical species of the habitat's communities and

ii. its implementation as a specific pressure indicator by including pressure-sensitive species.

Although the MSFD uses the term 'typical species' it neither provides a definition of this term, nor gives a list of typical species per habitat type. It is left to Member States to define lists of typical species and to set targets for their presence. For baseline setting (following OSPAR, 2011), the use of method C (current state) might be inappropriate if the habitats are exposed to high human pressure and no reference sites are available. Reaching a ratio of typical and/or character species similar to baseline conditions defined as past state of communities (i.e. method B) is more appropriate.

Many present-day studies have been undertaken in what is already a considerably altered environment. Data gathered prior to the era of intensive bottom fishing are sparse and therefore it is often difficult to deduce the original status of the benthos. This greatly hampers efforts to predict the outcome of management measures for existing benthic communities and habitats (Kaiser, 2003).

Bottom trawling is a source of chronic and widespread disturbance in shallow shelf seas and modifies the diversity, community structure, trophic structure and productivity of benthic communities (de Groot and Lindeboom, 1994; Dayton *et al.*, 1995; Jennings and Kaiser, 1998; Lindeboom and de Groot, 1998; Hall, 1999; Collie *et al.*, 2000; Kaiser and de Groot, 2000). The susceptibility of species is determined by their body size and turn-over rate rather than their taxonomic affiliation with large, slowly reproducing habitat-forming (e.g. horse mussels, see below) species being more susceptible than their smaller, faster reproducing (free-living) competitors (Kaiser *et al.* 2000; Jennings *et al.* 2001).

In a meta-analysis of 39 published fishing impact studies, Collie *et al.* (2000) found that the magnitude of the immediate response (i.e. change in abundance or biomass) of benthic organisms to fishing disturbance varied significantly according to the type of fishing gear used, the habitat and among different taxa. The most consistently interpretable result was with respect to faunal vulnerability, with a ranking of initial impacts that is broadly congruent with expectations based on morphology and behaviour.

Collie *et al.*'s (2000) regression tree analysis (Figure 4) provides a compelling quantitative basis for predicting the relative impacts of fishing under different situations. Following the tree from its root to the branches, predictions can be made, for example, about how a particular taxon would be affected initially by disturbance from a particular fishing gear in a particular habitat. Thus, trawling would reduce anthozoa (anemones, soft corals, sea ferns) by 68 %, whereas asteroid starfishes would only be reduced by 21 %. Similarly, repeated (chronic) dredging is predicted to lead to 93 % reductions for anthozoa, malacostraca (shrimps and prawns), ophiuroidea (brittlestars) and polychaeta (bristle worms), whereas a single (acute) dredge event is predicted to lead to a 76 % reduction.

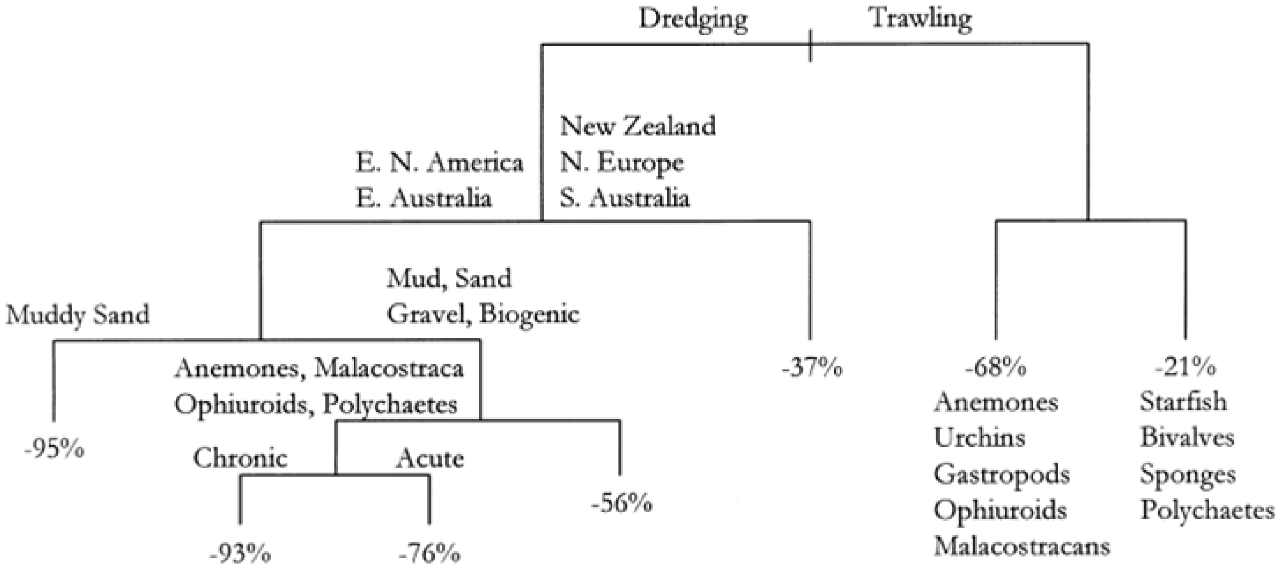

Figure 4. Regression tree of the initial response data. A binary partitioning algorithm recursively splits the data in each node until the node is homogeneous or contains too few observations. The vertical height of each branch indicates the importance of that split. The number under each node is the mean response for that combination of variables (after Collie *et al.* 2000).

Polychaetes were more negatively affected than oligochaetes, which appeared to be the least sensitive class. The most negative responses were for the polychaete species *Arenicola*, *Scoloplos*, *Heteromastus* and *Glycera*. Bivalves appeared to be less sensitive to fishing disturbance than gastropod molluscs. The bivalves *Macoma* and *Cerastoderma* were also more negatively impacted than the other genera, perhaps because they are specifically targeted by some fishing gears. The genera least impacted by disturbance were bivalves: *Nucula*, *Ensis*, *Chamelea*, *Abra* and *Corbula*. The mean response for *Nucula* was slightly positive. Many of these bivalves are small in size or have particularly well armoured shells that protect them from physical damage. Interestingly, none of the predicted means was positive. Taxa differed in their response to disturbance, but on average, none increased in abundance.

Complementing Collie *et al.*'s (2000) meta-analysis with literature (de Groot and Lindeboom, 1994; Lindeboom and de Groot 1998) and internet (MarLin) accounts, it is possible to define benthic genera comprising a community in a given region as being (1) likely to increase, (2) likely to decrease, or (3) one for which no a priori prediction can be made regarding their response to fishing (as an example see Table 2, from Frid *et al.* (1999), for a benthic community off the Northumberland coast in North East England).

Table 2. Benthic taxa categorised *a priori*, based on literature accounts of their response to fishing (after Frid *et al.* 1999).

Predicted decrease		No prediction		Predicted increase	
Phylum	Genus	Phylum	Genus	Phylum	Genus
Annelida	*Ampharete*	Cnidaria	*Virgularia*	Annelida	*Capitella*
	Anobothrus	Crustacea	*Ampelisca*		*Chaetozone*
	Levinsenia		*Diastylis*		*Commensodorum*
	Minuspio		*Ericthonius*		*Diplocirrus*
	Myriochele		*Eudorella*		*Exogone*
	Owenia		*Harpinia*		*Glycera*
	Prionospio		*Leucon*		*Glycinde*
	Rhodine		*Photis*		*Goniada*
	Spionid	Mollusca	*Abra*		*Heteromastus*
	Spiophanes		*Chamelea*		*Lumbrineris*
	Synelmis		*Lucinoma*		*Magelona*
	Terebellidae		*Mysella*		*Mediomastus*
	Terrebellides		*Nuculoma*		*Nephtys*
	Thelepus		*Tellimya*		*Ophelina*
Echinodermata	*Echinocardium*		*Thyasira*		*Paramphinome*
Mollusca	*Acanthocardia*	Nemertea	Nemertea		*Pholoe*
Phoronida	*Phoronis*	Platyhelminthes	*Polycladia*		*Praxilella*
					Pseudeurythoe
					Scalibregma
					Tharyx
				Echinodermata	*Amphiura*
					Ophoruoidea
				Oligochaeta	Oligochaeta

The next logical step is to examine expected changes in the proportion of benthic genera/species in response to fishing and compare these with changes that are predicted as a result of warming sea temperatures. If, for example, a benthic community is dominated by species/genera that are susceptible to both fishing and climate change, then positive trends-based targets for such species would be difficult, if not impossible, to meet.

Dynamic ecosystems and changing climates will lead to continuous changes in species composition and their relative abundance within communities and ecosystems in any given part of a region. So setting GES in a manner which is too specific in terms of the species composition and population sizes to be achieved will not allow for ecosystem changes (such as changing predator-prey relationships) or climatic variation. As these aspects are beyond the control of normal management measures, it could lead to GES and state targets being set in an unrealistic manner. Desired status should therefore be considered at the broader level of genera or groups of species, within which a suitable degree of fluctuation in species composition and relative abundance can be anticipated.

The long-term persistence of benthic species, their populations and communities in the face of climate change depends on their ability to keep pace with moving climates or adapt to changes *in situ*. Shifts in the distributional ranges of benthic populations in response to climate change have been observed (see section x), but it is less clear whether these shifts allow species to keep pace with climate change (Hiddink *et al.*, 2015). If species are not shifting at the trailing edge of their distribution (warm boundary) at the rate dictated by climate change, they will build-up an 'extinction' debt. Alternatively, species could adapt. Shifts at the leading edge (the cold boundary) are related to the ability of a species to disperse into areas that were previously too cold, and are likely to relate to the dispersal and settling capability of the species. A failure to colonise newly available habitat at the leading edge will result in an 'immigration lag' (Hiddink *et al.*, 2015 and references therein). Hiddink *et al.* (2015) used data from 65 benthic invertebrates occurring in the North Sea between 1986 and 2000 to assess whether organisms are shifting their distribution in response to ocean warming induced by climate change. Most species shifted their centre of the distribution at a rate that was slower than the shift in temperature. Over 32 % of species effectively did not shift their geographical distribution. Many of these species were thus experiencing sea bottom and surface temperatures in 2000 that was higher than in 1986 (Table 3).

Table 3. The percentage of species (*n* = 65) that are responding to climate change. 'Static and lagging': Species experiencing sea bottom temperatures (SBT) and sea surface temperatures (SST) in 2000 that were higher than in 1986; 'Shifting against temperature' = Species ending up in warmer waters than those they would have experienced if they had remained stationary; 'Shifting and tracking' = Species tracking temperature shifts; 'Shifting and over-compensating' = Species ending up at lower temperatures in 2000 than in 1986 (Hiddink *et al.*, 2005).

Temperature		Static and tracking	Static and lagging	Shifting against temperature	Shifting and tracking	Shifting and over-compensating
SBT	Minimum	2 %	49 %	28 %	15 %	6 %
	Mean	31 %	25 %	8 %	26 %	11 %
	Maximum	3 %	29 %	11 %	25 %	32 %
SST	Minimum	2 %	51 %	23 %	18 %	6 %
	Mean	6 %	78 %	0 %	15 %	0 %
	Maximum	6 %	55 %	8 %	18 %	12 %

It is likely that parts of those benthic invertebrate populations that are lagging behind temperature changes currently occupy unsuitable thermal habitat (Hiddink *et al.*, 2015). Changes in survival and reproduction of benthic species are thus possible. It is often assumed that stochastic variation in fertilisation success, planktonic duration or mortality rates will destroy any relationship between stock size, the production of larvae and the number of propagules still alive at the settlement. Hughes *et al.* (2000) argue that this assumption is not valid at larger scales where global recruitment must diminish if stock sizes are suppressed, e.g. as a result of climate change. Hughes *et al.* (2000) demonstrate that large-scale variation (~ 1800 km) in the density of coral recruits on the Great Barrier Reef, Australia, is associated with spatial and temporal changes in the fecundity of adults rather than changes in their abundance (i.e. where fecundity increased in their study, so did the density of recruits).

When setting targets for benthic habitat indicators, there is an underlying assumption that reductions in the size of benthic populations as a result of manageable human activities (e.g. fishing) are readily reversible because of an unlimited supply of recruits. However, large-scale degradation of adult breeding stocks, as a result of climate-driven environmental changes, could potentially prevent populations from recovering to baseline conditions. This could lead to recruitment failure in areas that are in most need to replenishment.

Multi-metric indices (BH-2)

Multi-metric indices consider diversity and abundance as well as the relative proportions of disturbance-sensitive, tolerant and opportunistic taxa. Many of the experimental studies investigating the effects of fishing on benthic communities have demonstrated a reduction in the abundance of species deemed to be vulnerable and in some cases species considered to be resilient have increased in abundance (Langton and Robinson, 1990; Tuck *et al.*, 1998; Frid *et al.*, 1999; Bergman and Van Santbrink, 2000; Bergman and Moore, 2001). Such changes in species abundance can result in reduced species richness and diversity (Kaiser and Spencer, 1996; Collie *et al.*, 1997).

Despite some suggestive patterns in the responses of number of individuals and species to fishing disturbance, few studies provide robust evidence on statistically significant effects of fishing on multi-metric indices. This lack of significance is largely due to the low statistical power of many studies and meta-analyses, but it may also be that negative responses (i.e. declines) of some species are counteracted by positive responses (i.e. increases) of others.

Whilst long-term changes in benthic communities may well be a fishing effect, an environmental influence cannot be discounted. Consequently, any relevant quantitative target for multi-metric indices will have to take account of the diversity, species richness and sensitive, tolerant, opportunistic species indicators used as well as the natural variability of species composition in space and time.

Working north of the island of Norderney in the German Bight where fishing pressure is among the lowest for the southern North Sea, Kröncke and Reiss (2010) examined the variability of multi-metric indices under environmental conditions prevailing over a 28-year period. All indices tested responded to natural disturbance events such as cold winters with the most pronounced effects after the severe winter 1978/79. As a result the ecological quality status changed from ‘good’ or ‘high’ ecological condition to ‘moderate’, ‘poor’ or even ‘bad’ conditions. However, not all indices responded to the disturbances in the same way and the strength of response varied remarkably between indices and the different disturbance events (i.e. gradual change in response to climate change versus drastic change in response to severe weather events which can lead to a decline of the corresponding ecological quality status across two to three classification units, see below).

The number of species and Shannon-Wiener Index, for example, showed a strong inter-annual variability (Figure 5). After 1986, during a period of increasing North Atlantic Oscillation Index (NAOI) and mild winters, the diversity index increased. During cold winters in 1995/96 and 1996/97, it decreased. After these winters, the index recovered and reached values as high as during the early 1990s. The notable increase of the Shannon-Wiener Index from the mid-1980s onwards coincides with a general ‘regime shift’ in faunal characteristics of the North Sea ecosystem in the mid- to late 1980s. Community changes were associated with increases in the NAOI and, linked to this, the influential role of water temperature. Mild meteorological conditions combined with a rising NAOI have resulted in an increase of macrofauna abundance, species number and biomass since 1988.

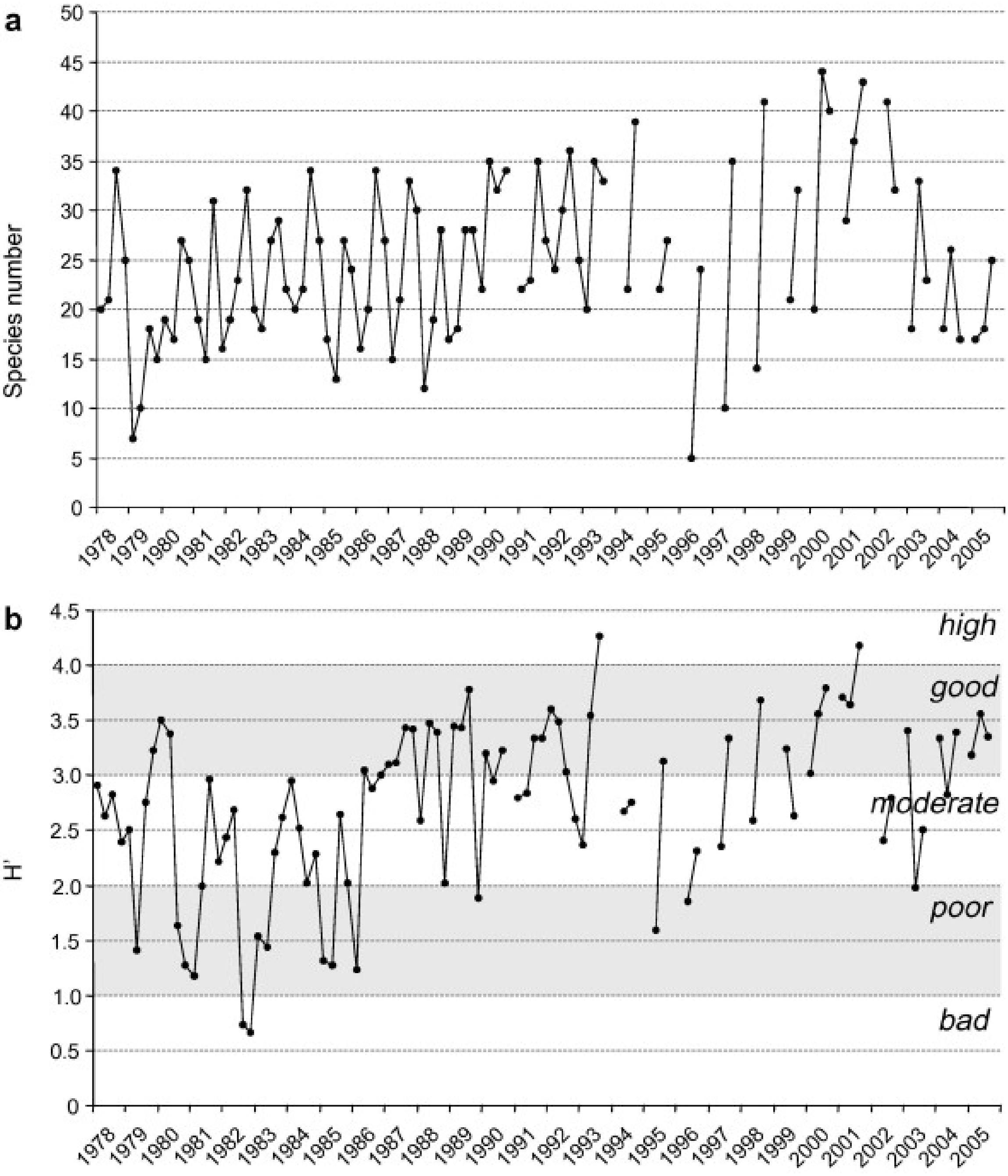

Figure 5. Variation of (a) Number of species and (b) Shannon-Wiener Index over time (1978–2005; after Kröncke and Reiss, 2010). The classification of the ecological status is based on Labrune *et al.* (2006).

Results for other multi-metric indices, such as the AZTI Marine Biotic Index (AMBI, based on the classification of benthic species into disturbance sensitive and tolerant species) and the Benthic Opportunistic Polychaetes Amphipods Index (BOPA, comparing percentage ratios of opportunistic polychaetes and amphipods) resembled a similar pattern. The mean AMBI of 1 to 2 reflects a 'high' to 'good' quality status, with the exception of the cold winter 1978/79, when it dropped to 'moderate' (> 3.3) mainly due to the dominance of the opportunistic polychaete species *Lagis koreni*, a species within the ecological group IV of AMBI (Figure 6). The BOPA increased notably in response to the cold winter of 1978/79 resulting in a shift of the corresponding ecological quality status from 'high' to 'poor' (Figure 6). Between 1980 and 2005, the index changed only slightly and remained in a 'high' quality status.

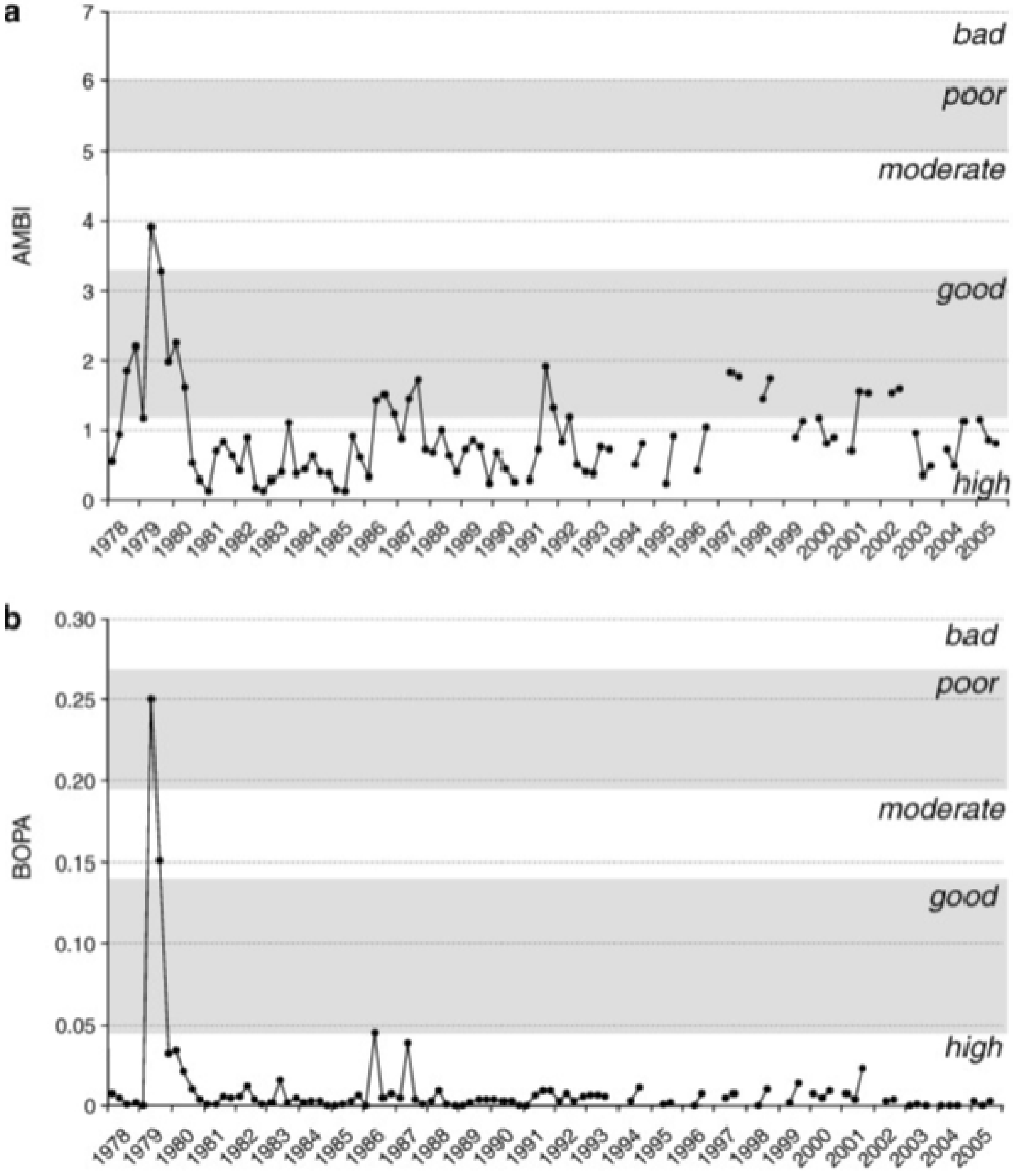

Figure 6. Variation of (a) AMBI and (b) BOPA over time (1978–2005, after Kröncke and Reiss, 2010). Classification of the ecological status for AMBI: high < 1.2, good = 1.2 - 3.3, moderate = 3.3 - 5.0, poor = 5.0 - 6.0, bad > 6.0; BOPA: high < 0.046, good = 0.046–0.140, moderate = 0.140–0.194, poor = 0.194–0.268, bad: 0.268–0.301).

Gradual, climate-induced long-term changes in species diversity since the late 1980s resulted in an increase of the ecological quality status from 'moderate' to 'good' for multimetric indices based on the abundance and distribution of all species (e.g. Shannon-Wiener Index). In contrast, multi-metric indices, which are mainly based on the ecological grouping of species (e.g. AMBI), did not reflect these gradual changes of the benthic community. Kröncke and Reiss' (2010) study provides compelling evidence that climate-induced community changes over time have to be considered when setting reference conditions for benthic ecosystems. Benthic communities may have reached a new stable state so that efforts towards a retrogression of community parameters, and thus ecological status, might be inappropriate.

Physical damage of predominant and special habitats (BH-3)

Benthic habitat types are very varied across NE Atlantic waters, ranging from widespread habitat types (e.g. sublittoral sand and mud) to those habitats which tend to be spatially discrete and historically more vulnerable to human pressures (e.g. biogenic reef). The MSFD treats habitat in the same way as EUNIS where the term habitat addresses both the abiotic characteristics and the associated biological community.

A number of benthic habitats are listed for protection under a variety of national and international obligations (e.g. EC Habitats Directive and the OSPAR Convention). These habitats are referred to as 'special habitats' under the MSFD due to their special scientific or biodiversity interest. These special habitats are of high conservation importance, but do not cover the full diversity of benthic habitats. The majority of the seabed in NE Atlantic waters consists of what the MSFD terms 'predominant habitat types' which are the less spatially discrete, non-listed habitats (Table 4). Ecological knowledge of predominant habitats is more limited than that of the special (listed) habitats. Hence, these habitats are described at a generic level based on geophysical descriptors (e.g. depth and substratum).

Table 4. OSPAR list of 'Predominant' and 'special' benthic habitat types in NE Atlantic waters

Special benthic habitat	Predominant benthic habitat
Annual vegetation of drift lines	Littoral rock and biogenic reef
Atlantic salt meadows	Littoral sediment
Intertidal *Mytilus edulis* beds on mixed and sandy sediments	Shallow sublittoral rock and biogenic reef
Carbonate mounds and associated communities	Shallow sublittoral coarse sediment
Coastal/saline lagoons	Shallow sublittoral sand
Cold water coral reefs/*Lophelia pertusa* reefs	Shallow sublittoral mud
Coral gardens	Shallow sublittoral mixed sediment
Deep-sea sponge aggregations/communities	Shelf sublittoral rock and biogenic reef
Horse mussel (M*odiolus modiolus)* beds	Shelf sublittoral coarse sediment
Intertidal chalk/littoral chalk communities	Shelf sublittoral sand
Intertidal mudflats	Shelf sublittoral mud
Large shallow inlets and bays	Shelf sublittoral mixed sediment
	Upper bathyal rock and biogenic reef
	Upper bathyal sediment
	Lower bathyal rock and biogenic reef
	Lower bathyal sediment
	Abyssal rock and biogenic reef
	Abyssal sediment

The BH-3 indicator aims to address pressures causing physical damage to seafloor habitats. It is designed to assess predominant as well as special habitat types and is regarded as particularly useful to target larger sea areas with relatively low effort. The indicator builds upon two types of information

i. the distribution and sensitivity of habitats and

ii. the distribution and intensity of human activities that potentially cause physical damage, such as mobile bottom gear fisheries.

For each habitat type, a baseline of the area and degree of damage has to be determined, as well as the natural extent of the habitat type. A proposed target of 15 % of the baseline value has been endorsed by OSPAR and HELCOM. This target level was similarly proposed by HELCOM. For special habitats a target of 5 % or less was recommended previously. Under the Habitats Directive if more than 25 % of the extent of the habitat is damaged (specific structures and functions including typical species) it is classed as 'unfavourable-bad'.

Auster and Langton (1998) reviewed a wide range of studies that reported effects of fishing on habitat (i.e., structural habitat components, community structure, and ecosystem processes) for a diversity of habitats and fishing gear types. Commonalities of all studies included immediate effects on species composition and diversity (see BH-1 above) and a reduction in habitat complexity. Studies of acute effects were found to be a good predictor of chronic effects. Structurally complex habitats (e.g. biogenic reefs) and those that are relatively undisturbed by natural perturbations (e.g. deep-water mud substrata) are more adversely affected by fishing than unconsolidated sediment habitats that occur in shallow coastal waters. Structurally complex and stable habitats also have the longest recovery trajectories in terms of the re-colonization of the habitat by the associated fauna (Kaiser, 2003).

M. modiolus forms dense beds, at depths up to 70 m (but may extend onto the lower shore), mostly in fully saline conditions and often in tide-swept areas. Although *M.modiolus* is a widespread and common species, horse mussel beds (with typically 30% cover or more) are more limited in their distribution. *M.modiolus* beds are found on a range of substrata, from cobbles through to muddy gravels and sands, where they tend to have a stabilising effect, due to the production of byssal threads. Communities associated with *M.modiolus* beds are diverse, with a wide range of epibiota and infauna being recorded, including hydroids, red seaweeds, solitary ascidians and bivalves such as *Aequipecten opercularis* and *Chlamys varia*.

M. modiolus reefs and their associated epifaunal communities have been found to decline in areas subjected to bottom-towed fishing gear. Some of the most detailed accounts are from the United Kingdom where scallop dredging and trawling have resulted in a reduction in quality and extent of historical *M. modiolus* dominated habitat (Service and Magorrian, 1997; Bradshaw *et al.*, 2002; Strain *et al.*, 2012). Strangford Lough in the Irish Sea is to date the best case study of the effects of bottom trawling on *M. modiolus* communities. Holt *et al.* (1998) summarised findings from various studies on the dynamics and sensitivity of *M. modiolus* beds to fishing pressure and climate change. Towed fishing gear and especially scallop dredges and heavy beam trawls tear up clumps of mussel. Even if individual mussels survive fishing impact, the structural integrity of the biogenic reef will have been disrupted and the ecosystem degraded through damage to the associated biota. The remaining reef may be subject to erosion or to greater abrasion from sand and shell which is no longer stabilised by the byssal threads. Repeated fishing episodes generally result in the replacement of complex beds with small clumps of living mussels amongst masses of dead shells. Factors affecting the length of time taken for the regeneration of biogenic reefs following reduction in fishing pressure include the period of non-disturbance, proximity of propagule sources and hydrodynamic influences on propagule dispersal. In addition, suitable habitat for keystone species has to be available for the regeneration process to start (Elsäßer *et al.*, 2013 and references therein).

M. modiolus is a long-lived mussel of circumboreal distribution and dense aggregations seem to reach their southerly limit around British shores. Climate change is forecast to inhibit recruitment of *M. modiolus* at the southern limits while permitting colonisation of parts of the Arctic, such as Svalbard, where the species is known from subfossil material to have occurred during a period in the early Holocene when sea temperatures were higher. There are locations suitable for long-term expansion in the north to balance losses in the south (Holt *et al.*, 1998). Owing to the long life-span of *M. modiolus*, there will be a considerable time-lag between climate shift and observable habitat changes. Failures of recruitment or of the survival of young mussels through the period when they are most vulnerable to predation (Seed and Brown, 1978) are the most likely ways change will come about in the south. Having shown an ability to colonise artificial habitats offshore (Fariñas-Franco *et al.*, 2013), it is likely that where suitable habitat exist, *M. modiolus* will be able to spread further north. However, warmer seas may prevent recovery of *Modiolus* beds damaged by fishing and/or recruitment of juveniles to undamaged beds. It thus follows that a recovery to a footprint of damaged *M. modiolus* habitat of < 5 % might be impossible, at least in the south of their distributional range, despite reduction in fishing pressure.

Area of habitat loss (BH-4)

The BH-4 indicator assesses the proportion of the area of habitats that are lost permanently or for a long-lasting period due to anthropogenic pressures. In special habitats which are defined by long-lived habitat-bio-engineering species, such as reefs of the cold-water coral *Lophelia pertusa* or the horse mussel *Modiolus modiolus*, changes in the extent of the habitat are likely to be due to anthropogenic physical influences as exerted by bottom-trawl fisheries (Lindenbaum *et al.* 2008; Strain *et al.* 2012). Loss of extent of habitat is of most concern for these biogenically-defined habitats and others including seagrass beds (Godet *et al.*, 2008). In general, habitat loss is less of an issue for physically-defined special habitats such as rock reef or large shallow inlets and bays, because these habitats are typically less spatially sensitive and have a greater natural extent. As a target value, damaged or lost area per predominant habitats should not exceed 15 % of the baseline value.

Gormley *et al.* (2013) used publicly available datasets to model the extent of habitat suitable for *M. modiolus* beds around the UK coastline under current conditions and to predict habitat loss under an increased climate change scenario. Their results indicated that there would be a decrease of potentially suitable habitat by 2050 (58% loss by 2030; and 98% loss by 2050) and complete loss of suitable *M. modiolus* bed habitat by 2080 (Figures 7 and 8).

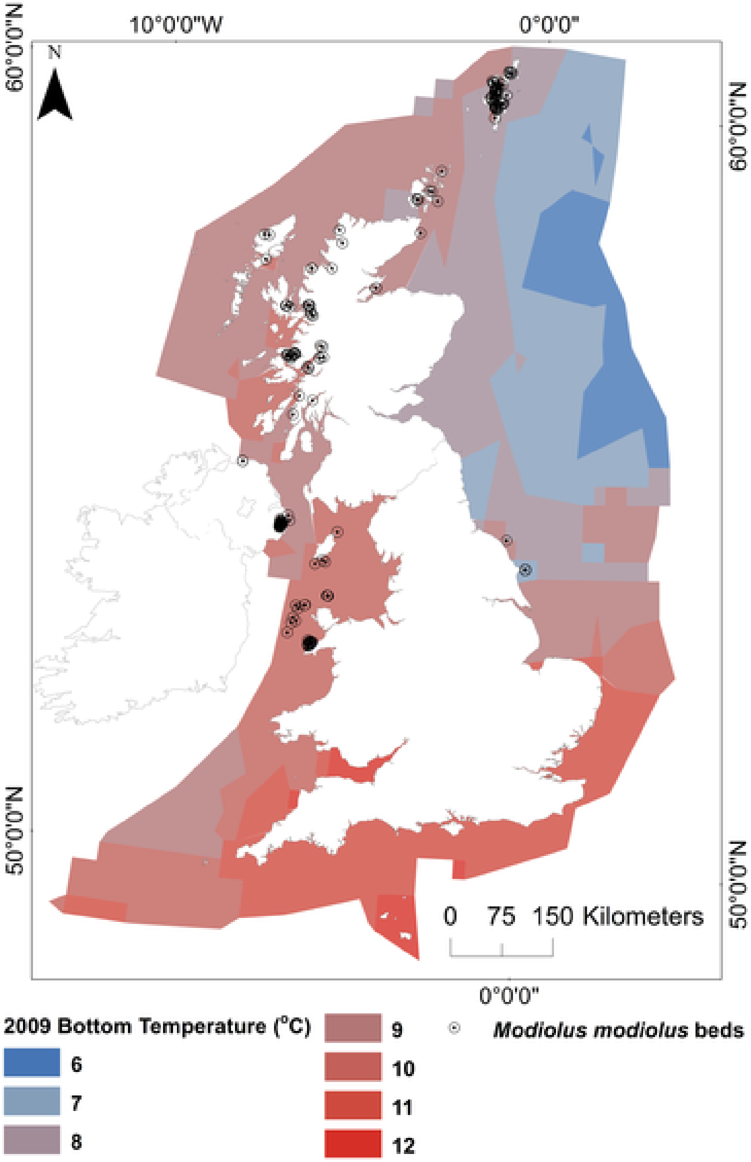

Figure 7. Current known distribution of M. modiolus beds around the UK coastline and illustrated 2009 baseline seabed temperature (Gormley *et al.*, 2013)

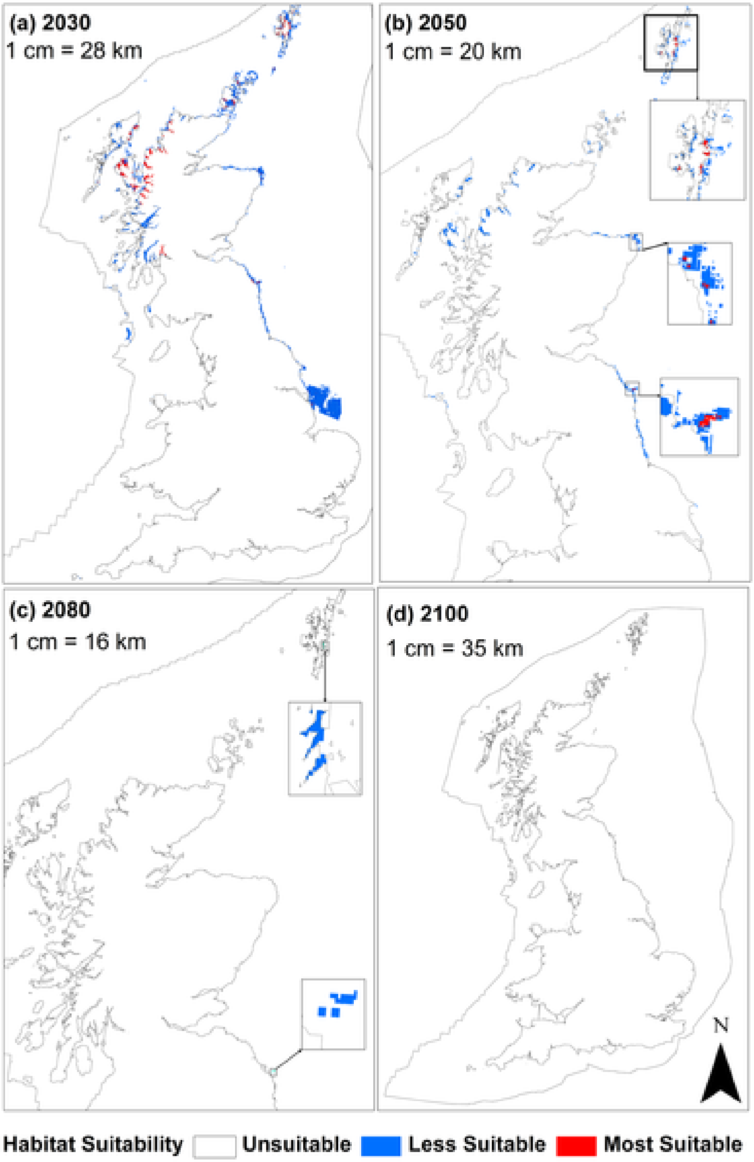

Figure 8. Model prediction maps for M. modiolus beds around the UK coast line for four projected climate change epochs (a) 2030, (b) 2050, (c) 2080, (d) 2100 (Gormley *et al.*, 2013).

Presently, UK GES targets under the MSFD for rock and biogenic reefs are drawn from the Habitats Directive i.e. that the "Area is stable or increasing and not smaller than the baseline value" (EU Habitats and Species Directive, Council Directive 92/43EEC). This is in keeping with one of the key aims of the MSFD to "Protect and preserve the marine environment prevent its deterioration or, where practicable, restore marine ecosystems". However, one of the key MSFD characteristics of D1 is that "The quality and occurrence of habitats and the distribution and abundance of species are in line with prevailing physiographic, geographic and climatic conditions". Gormley *et al.*'s (2013) model predictions suggest that, maintaining nationally stable or increasing areas of *Modiolus* beds may not be achievable within the next 40 years by reducing fishing pressure alone.

Size-frequency distribution of bivalve or other sensitive/indicator species (BH-5)

The natural balance between both, the large and small species within benthic communities and the large and small specimens within populations of a single species can be affected by anthropogenic influences such as physical disturbance (e.g. caused by bottom trawling). As the community-based approach is partly covered by other indicators (e.g. typical species composition, multimetric indices, see above), this section focuses on the population structure of single indicator species. This is the only proposed condition targeted benthic habitats indicator with a focussed sensitivity to physical disturbance. It is therefore considered suitable not only for regular state assessment but also for monitoring of the success of measures – provided meaningful GES targets can be set.

Baselines have not yet been defined and will depend on habitat type. The use of reference state as a baseline is regarded to be the most appropriate approach. The reference level is a natural population structure of the selected species with a defined proportion of small (young), medium-sized and large (old) individuals. This reference not only strongly depends on the species, but also on parameters with high spatial variability like spawning success, growth rate and mortality. The 'natural' size distribution of a population is very unlikely to be found in the field as most areas are already impacted. Additionally, the non-impacted area has to be environmentally comparable to the impacted area. Historical data are also very rare as size measurements have not been part of the standardised protocol in most monitoring programmes. Most probably, the natural condition of the population will therefore have to be defined using population dynamic models which have, up to now, rarely been applied in marine benthic ecology.

In principal, BH-5 is based on the number of individuals (or their biomass) per size class. In general, size is strongly correlated with age and hence the size-frequency structure of a population can give valuable insights into the severity, frequency and longevity of impacts. The impact of human pressure on the size-frequency distribution of single species has been described primarily for fish species whereas regional studies on benthic invertebrates are much rarer.

Large bivalves such as *Arctica islandica* are regarded to be a sensitive group of benthic species to bottom-trawling (Witbaard and Bergmann, 2003). *A. islandica* is native to the North Atlantic Ocean and has a general distribution around all British and Irish coasts and offshore. The species has a sub-tidal distribution and typically presents on firm coarse substrata, usually fully or partially buried in fine to coarse grained sands. *A. islandica* is exceptionally long-lived with individuals at the southern limits of its range reaching ages of several hundreds of years old. Consequently, *A. islandica* has become a species of significant academic interest for climate reconstruction over the last two decades (e.g. Butler *et al.*, 2013). Population structure in the North Sea has been examined through the work of Whitbaard and Bergman (2003) and Ridgeway *et al.* (2012) looked at the population structure in outer Belfast Lough and reported on the estimated longevity and natural mortality rates for the species in this area.

Papers on growth of *A. islandica* from the North Sea (Witbaard and Duineveld, 1990), Baltic Sea (Brey *et al.*, 1990) and Kattegat (Josefson *et al.*, 1995) indicate considerable differences in the growth rates of *A. islandica* in north-west European waters. Shell growth itself has been shown to be modified directly by bottom water temperature and food supply. Harding *et al.* (2008) examined long-term recruitment patterns of *A. islandica* in the Mid-Atlantic Bight (MAB) off the east coast of North America across temporal (decadal) and spatial (latitudinal, bathymetric) scales. Recruitment was high in years in which the number of months with water temperatures averaging 6°C to 10°C exceeded the number of months with water temperatures less than 6°C by at least two months. In general, years with above average bottom water temperatures during January, February, and March tended to produce strong year classes. While mature *A. islandica* can withstand temperatures up to 20°C (Loosanoff, 1953), such temperatures are detrimental to larvae. Lutz *et al.* (1982) observed optimum growth rates between 13 and 15°C in the laboratory. At higher temperatures, larval growth decreases and ultimately stops. This is consistent with the southern limit of *A. islandica*'s distribution in the North Sea which coincides with the summer stratified water mass where bottom water temperatures never exceed 16°C (Witbaard and Bergman, 2003).

The long-term persistence of a 'natural' population structure of fishing-sensitive species such as *A. islandica* in the face of warming seas will greatly depend on successful recruitment. Studying the distribution, abundance and population structure of *A. islandica* in the North Sea between 1970 and 2000, Whitbaard and Bergman (2003) observed that the recruitment to larger size classes was hampered in the Oyster ground. An obvious explanation for the low abundance of spat is a low density of reproductively active adults. Low adult densities would lower the fertilisation success and thus the number of available larvae. One of the factors that have controlled the size of the adult stock in the Oyster Ground over the last decades is the intensive beam trawl fisheries for flatfish. Direct mortality of adults due to physical damage would results in insufficiently dense stocks of reproducing adults generating less dense spatfalls. Although fishing may be a major reason for the skewed population structure, climate-change-induced increases in water temperatures, leading to reduced survival of spat and juveniles, could be a contributing factor as well. It is therefore questionable whether reducing fishing pressure alone will be sufficient to ensure sustainable populations of *A. islandica* in the SE North in the long term.

Fish Communities

Population biomass/abundance of a suite of sensitive species (1.2.1, FC-1)

Species level indicators of abundance and/or biomass are required to support implementation of the MSFD in respect of Criterion 1.2 (population size) of Descriptor 1 "Biological diversity is maintained". Greenstreet *et al.* (2012) proposed an indicator and assessment process to address this need in respect of a suite of fish species chosen by virtue of their 'slow-type' life-history traits (large ultimate body size, slow growth rate, late age at maturity, large length at maturity). Fish species characterised by such traits are considered to be particularly vulnerable to fishing mortality (Jennings *et al.*, 1998; Gislason *et al.*, 2008; Le Quesne and Jennings, 2012). Greenstreet *et al.* (2012) trialled their approach in the North Sea using first quarter (Q1) international bottom trawl survey (IBTS) data. Over the period 1983 to 2008 for which Q1 IBTS data were analysed, 119 demersal species were sampled. These were ranked according to their life-history traits and the 40 species (33%) with the 'slowest-type' life-history traits were labelled 'sensitive' species. Data for 13 species were inadequate to support assessment but, in terms of their range of life-history traits, the remaining 27 species were considered representative of the full suite of 40 'sensitive' species. Fishing pressure on North Sea fish increased considerably during the 20th century (Thurstan *et al.*, 2010) and populations of 'slow-type' life history trait species declined markedly (Walker and Hislop, 1998; Jennings *et al.*, 1999; Greenstreet and Rogers, 2000; van Strien *et al.*, 2009). Greenstreet *et al.* (2012) therefore argued that by 2008, human activities in the North Sea had depressed populations of their suite of 27 'sensitive' species. Consequently, population recovery would be necessary to achieve good environmental status, requiring the setting of positive trends-based targets for population abundance/biomass metrics.

Abundance/biomass trends for most of the 27 'sensitive' species were non-monotonic so Greenstreet *et al.* (2012) chose to set non-parametric trends-based targets, requiring that abundance/biomass metric values at the time of assessment should fall within the upper 25%ile of all metric values observed within the full time-series of values observed for each species. Using a Brownian random walk simulation, the probability of a species meeting such a target simply by chance was assessed at $p = 0.332$. Given that abundance/biomass trends for 27 species were analysed, each with this probability of meeting the target by chance, the binomial distribution suggests that observing 14 species meeting their targets would occur with a probability of $p < 0.05$. In this case study therefore, the species level indicator target was set as *at least 14 species among the defined suite of 27 North Sea 'sensitive' demersal fish species should meet individual trends-targets such that, at the time of assessment, prevailing abundance/biomass metric values lie within the upper 25%ile of all metric values observed within the full time-series of values observed for each species*.

However, a key assumption underlying the setting of such a target is that each of the 27 'sensitive' species assessed does indeed have the same random probability of $p = 0.332$ of meeting its own trends-based target. The review above shows that fish communities in the Northeast Atlantic have been profoundly influenced by changes in the marine environment and the consequent knock-on effects mediated through marine food webs. Rising sea temperatures have caused shifts in species distributions, resulting in increases in the abundance/biomass of southern-affinity species within marine regions such as the North Sea, and potential declines in the abundance of northern-affinity Boreal species. A key question then concerns the biogeographic affinity of species selected in suites of 'sensitive' species for different MSFD sub-regions. Recent environmental change might be expected to encourage recovery among southern-affinity Lusitanian and Atlantic species and to suppress recovery among Boreal species (Engelhard *et al.*, 2011). If suites of species for MSFD subregions such as the Greater North Sea include a high proportion of Boreal species, this would make achieving the species-level indicator targets, such as the one stated above, much more difficult to achieve. In fact, should the proportion of Boreal species within the suite be sufficiently high, and if these species were actually to decline in response to continuing warming of the seas, then meeting such targets could ultimately prove impossible. To explore the potential risk of such a situation occurring, the North Sea Q1 IBTS data analysed by Greenstreet *et al.* (2012) were re-evaluated taking account of the biogeographic affinity of each of the 27 'sensitive' species.

Table 5 provides a list of the 27 demersal fish species making up the North Sea suite of 'sensitive' species along with their biogeographic affinity. The table indicates whether each species either met or failed its own population biomass and population abundance trends-based targets in 2008; 11, or 12 species met their abundance, or biomass, targets respectively, both just short of the species level indicator target of 14 (Greenstreet *et al.*, 2012). However, trends for both metrics were encouraging; with extrapolation suggesting that the target might be met by both metrics by 2020 (Greenstreet *et al.*, 2012). However, when biogeographic affinity is taken into account, such optimism appears misplaced. The proportion of 'sensitive' species meeting the two population size targets was much higher among species of Lusitanian and Atlantic affinity than among species of Boreal origin (Figure 9). Furthermore the combined biomass, or abundance, of all 12 'sensitive' Boreal species greatly exceeded the combined biomass, or abundance, of the 15 'sensitive' Atlantic/Lusitanian species, and while increasing trends in both biomass and abundance were apparent in the latter, strong declining trends were evident in the former (Figure 10).

Table 5. Suite of 27 'sensitive' species analysed in North Sea case study (Greenstreet *et al.*, 2012). The biogeographic affinity of each species is indicated along with the results of 2008 assessment regarding whether each species met its own individual trends-based population abundance and population biomass targets.

Biogeographic affinity	Scientific name	Common name	Biomass	Abundance
Atlantic	*Helicolenus dactylopterus*	Bluemouth	Failing	Failing
Atlantic	*Anguilla anguilla*	European eel	Meeting	Meeting
Atlantic	*Chimaera monstrosa*	Rabbit ratfish	Meeting	Failing
Lusitanian	*Leucoraja naevus*	Cuckoo ray	Failing	Failing
Lusitanian	*Lophius piscatorius*	Angler	Failing	Failing
Lusitanian	*Mustelus asterias*	Starry smooth hound	Failing	Failing
Lusitanian	*Mustelus mustelus*	Smooth hound	Failing	Failing
Lusitanian	*Chelidonichthys lucerna*	Tub gurnard	Meeting	Meeting
Lusitanian	*Galeorhinus galeus*	Tope	Meeting	Failing
Lusitanian	*Lepidorhombus whiffiagonis*	Megrim	Meeting	Meeting
Lusitanian	*Merluccius merluccius*	Hake	Meeting	Meeting
Lusitanian	*Raja brachyura*	Blond ray	Meeting	Meeting
Lusitanian	*Raja clavata*	Thornback ray	Meeting	Meeting
Lusitanian	*Raja montagui*	Spotted ray	Meeting	Meeting
Lusitanian	*Scyliorhinus canicula*	Lesser spotted dogfish	Meeting	Meeting
Boreal	*Anarhichas lupus*	Catfish	Failing	Failing
Boreal	*Brosme brosme*	Torsk	Failing	Failing
Boreal	*Cyclopterus lumpus*	Lumpsucker	Failing	Failing
Boreal	*Dipturus batis*	Skate	Failing	Failing
Boreal	*Gadus morhua*	Cod	Failing	Failing
Boreal	*Molva molva*	Ling	Failing	Failing
Boreal	*Pollachius pollachius*	Pollack	Failing	Failing
Boreal	*Sebastes viviparus*	Norway haddock	Failing	Failing
Boreal	*Squalus acanthias*	Spurdog	Failing	Meeting
Boreal	*Zoarces viviparus*	Viviparous blenny	Failing	Failing
Boreal	*Hippoglossus hippoglossus*	Halibut	Meeting	Meeting
Boreal	*Pollachius virens*	Saithe	Meeting	Meeting

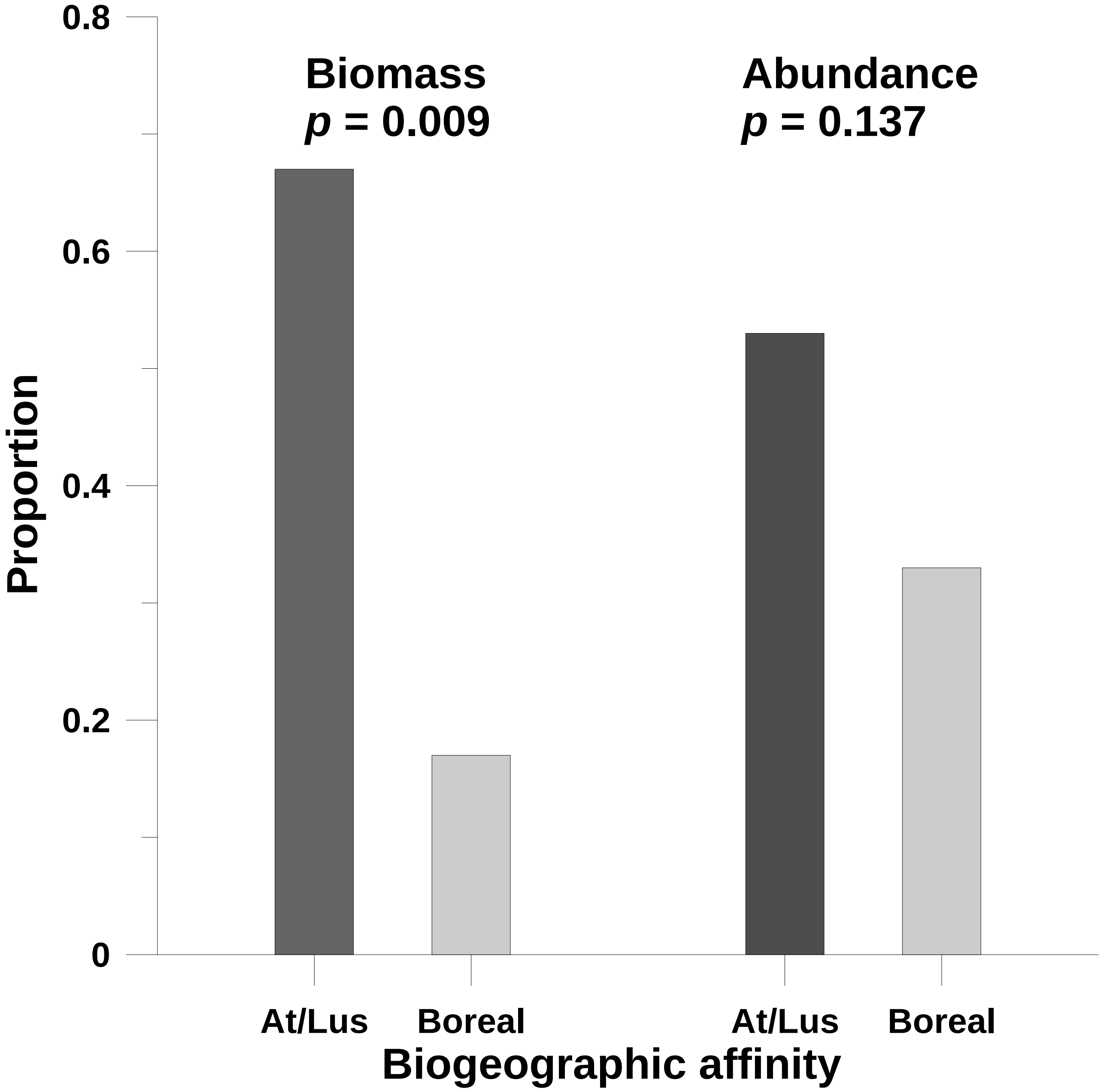

Figure 9. The proportion of 'sensitive' species meeting individual trends-based population abundance and population biomass targets, grouped into species with either Atlantic/Lusitanian (At/Lus) or Boreal biogeographic affinities. Significance of the differences between proportions was assessed using 2x2 contingency tables.

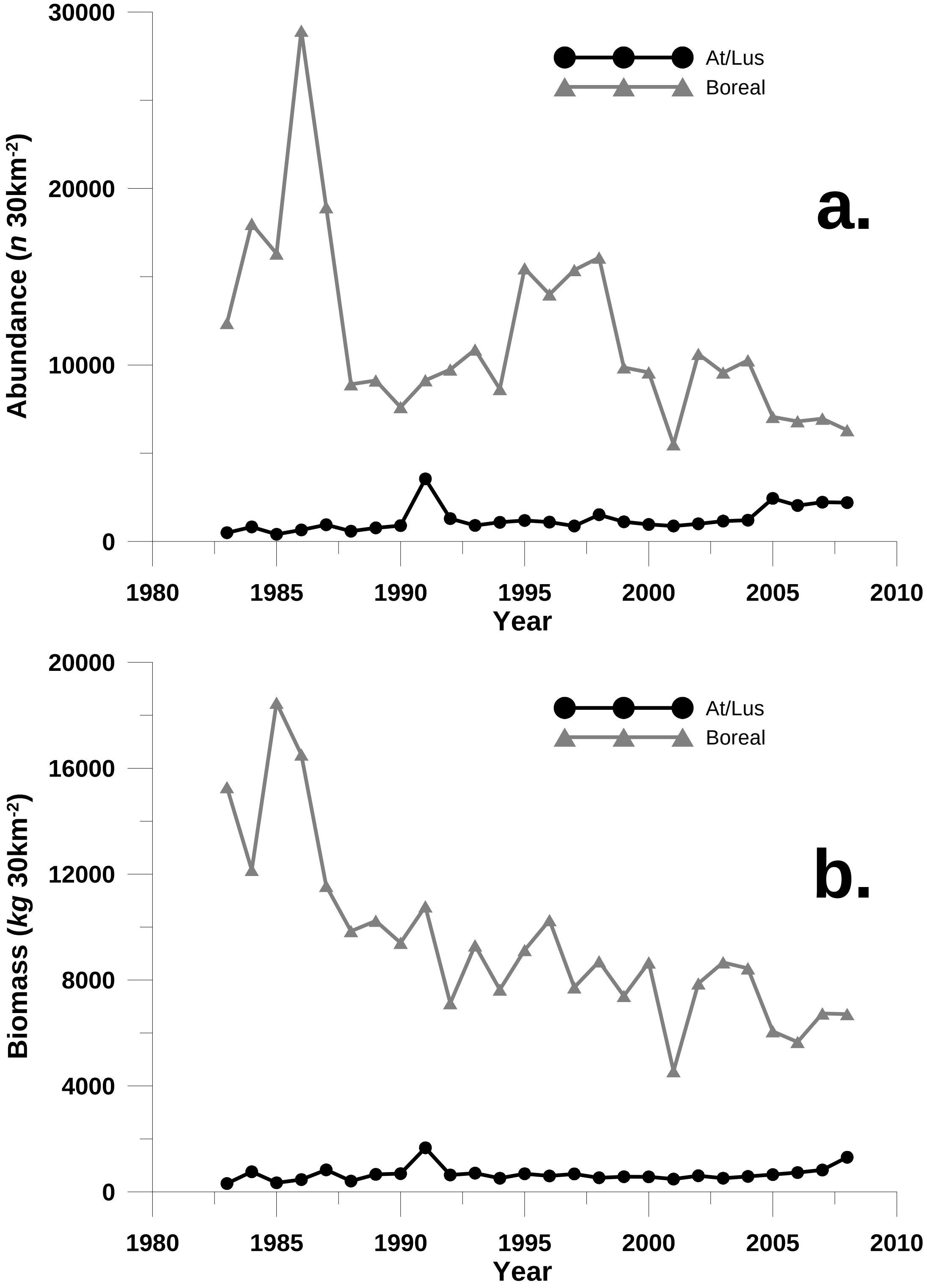

Figure 10. Trends in the total abundance (a.) and biomass (b.) of the suite of 27 'sensitive' demersal fish species in the North Sea with either an Atlantic/Lusitanian (At/Lus) or a Boreal biogeographic affinity.

It would seem that much of the increase in the number of 'sensitive' species meeting their individual abundance and biomass targets reported by Greenstreet *et al.*, (2012) was driven primarily by increases in the population size of species of Atlantic and Lusitanian affinity. Whilst reduced fishing mortality almost certainly helped facilitate these population expansions, it seems more than likely that the changes in environmental conditions that have occurred in the North Sea have also contributed considerably. Conversely, only two of the Boreal 'sensitive' species, halibut and saithe, met both abundance and biomass targets (the abundance target was also met by spurdog). Given the overall decline in total combined Boreal 'sensitive' species biomass and abundance, the prognosis for the remaining 10 Boreal species does not seem optimistic. There seems little doubt that excessive fishing mortality was responsible for their population declines in the first instance, but now that fishing pressure has been reduced (Greenstreet *et al.*, 2009; Greenstreet *et al.* 2011; Tidd, 2013) it would appear that the changing marine environment is inhibiting their recovery.

This has major implications with regard to meeting the overall target requiring 14 of the suite of 27 'sensitive' species to meet their own individual recovery-linked trends-based targets. If recovery among 10 of these species were to be inhibited because of changing environmental conditions, this would require 82% of the remaining 17 species to meet their targets; a much tougher requirement than the original 52% (14 out of 27). The overall target for the suite of 'sensitive' species, requiring 14 species to meet their own recovery-linked trends-based targets, might have to be reassessed if changes in "*prevailing physiographic, geographic and climate conditions*" are to be adequately taken account of. Perhaps the most depressing aspect of this re-evaluation of Greenstreet at al. (2012) is the observation that, despite nearly two decades of improvement in fisheries management in the North Sea, the overall biomass and abundance of 27 of the most 'sensitive' species in the demersal fish community continues to decline.

Relative proportion of ecosystem components – proportion of large fish (1.7.1, FC-2)

Application of the ICES (2001a) criteria for identifying good state indicators suggested that size-based indicators would most effective at monitoring the detrimental impact of fishing on fish communities (ICES 2001b; Greenstreet, 2008). This subsequently led to the development of the large fish indicator (LFI) to support establishment of an Ecological Quality Objective for the North Sea demersal fish community as part of the OSPAR pilot project to develop ecosystem based marine management (EBMM) (Greenstreet *et al.*, 2011). The MSFD requires MSs where appropriate to use indicators that have already been established and which already fulfil established functions, for example under Regional Seas Conventions (EC, 2008). The LFI monitors change in the fraction of total demersal fish biomass consisting of fish exceeding a specified 'large fish' threshold length. As such it monitors change in the relative proportions of 'large' and 'small' fish within demersal fish communities, rendering it suitable as a "*relative proportion of ecosystem components*" indicator (1.7.1 in the Decision document; EC, 2010). LFIs have been defined for the North Sea (Greenstreet *et al.* 2011), Celtic Sea (Shephard *et al.*, 2011) and southern Bay of Biscay (Modica *et al.*, 2014), so it has been widely adopted across the OSPAR area, which provides strong support for its adoption as a "common indicator" for assessing fish community status at MSFD regional and subregional scale.

A biodiversity perspective essentially underpinned the target setting process in all three studies cited above; all three studies attempted to define the state of the community at the point beyond which any further detrimental change caused by fishing would become unacceptable. Here we reassess the target set for the LFI in the North Sea, and we again consider this from the same biodiversity perspective: the 'health' of the North Sea demersal fish community. We make this point because, due to the profound size-structuring of marine food webs (Sheldon *et al.*, 1972; Kerr and Dickie, 2001; Jennings *et al.*, 2001; Jennings *et al.*, 2002; Duplisea, 2005), the LFI is also proposed as a food web indicator to fulfil the Decision document indicator 4.2.1 role (EC, 2010) and it is currently being considered as an OSPAR "common indicator" for food webs (FW-3). As yet however, the relevance of currently proposed LFI targets as appropriate targets when using the LFI within a food web context has still to be tested.

For the North Sea, the EcoQO has been set at a value of LFI ≥ 0.3, the observed LFI value in 1983, the first year in the Q1 IBTS time series used to derive the LFI. This is because ICES (2006a) considered the early 1980s to be "*the last period when ICES advice regarding the management of the exploited species was generally for the maintenance of status quo exploitation rates, suggesting that this was the last period when science experts considered fishing to be generally sustainable in the North Sea*". Greenstreet *et al.* (2011) subsequently showed that fishing mortality in 1983 was already excessive but, because of time-lags in the response of demersal fish populations to changes in fishing mortality, the state of the fish community in 1983 was still commensurate with the 'sustainable-use' state. Comparison with much longer-term Scottish August Groundfish Survey data (e.g. Greenstreet and Hall, 1996; Greenstreet and Rogers, 2006), suggested that an LFI of around 0.3 would have prevailed throughout much of the 20th century leading up to the early 1980s (Greenstreet *et al.*, 2011).

Greenstreet *et al.* (2011) observed a lagged LFI response to changes in fishing mortality with lags >12y, similar to those noted in an earlier study (Daan *et al.*, 2005). Applying these lagged relationships to recent fishing mortality data, suggested that the EcoQO of an LFI ≥ 0.3 might well be achieved by 2020. However, subsequent analysis has suggested that the rate of recovery in the North Sea LFI has slowed (Fung *et al.*, 2012), while modelling studies suggest that full recovery to the EcoQO level could take decades (Fung *et al.*, 2013), or might not be achievable under current environmental and management scenarios (Blanchard *et al.*, 2014). Nevertheless increases in the LFI since 2000 have been linked to declines in fishing effort of demsersal trawlers (Englehard *et al.* in press). Here we reanalyse an updated version of the Q1 IBTS data set to re-examine the lagged relationship between the LFI and fishing mortality. In particular, we examine whether the observed slowing of the rate of recovery in the LFI is predicted by this relationship to fishing mortality, or whether this might instead suggest an increase in the influence of environmental forcing. We also consider recent documented (see above) changes in the North Sea demersal fish community linked to changing environmental conditions to assess whether the current EcoQO target for the LFI of LFI ≥ 0.3 is still relevant under currently "*prevailing physiographic, geographic and climate conditions*".

The LFI time series was updated to include data to 2011. The community averaged fishing mortality indicator (F_{com}) of fishing pressure on the North Sea demersal fish community time-series was also reconstructed to take account of the most recent stock assessment estimates of fishing mortality as well as utilising recently constructed historic estimates. A time-lagged relationship between the LFI and F_{com} was still evident with significant (using the Chelton procedure to take account of autocorrelation in both time series) negative relationships apparent with lags from 14 to 20 years. The closest relationship ($LFI_y = 0.4696 - 0.2696F_{com,y-17}$, $r^2 = 0.556$, $N^* = 13$, $p = 0.002$) was obtained with a lag of 17y. Using this relationship applied to F_{com} data for the period 1947 to 2012, a predicted LFI trend could be generated spanning the years 1964 to 2029 (Figure 11). The observed LFI values for 1983 and 1985 stand out as reasonably substantial outliers from the modelled trend. Since the LFI target of 0.3 was based on the 1983 value, this perhaps suggests that the original target of 0.3 might have been too ambitious in the first place. However, earlier on in the modelled time series, between 1964 and 1977, the modelled LFI varied around a value of 0.28, suggesting that the original target of 0.3 may not be that unreasonable. Furthermore the modelled trend exceeds the target value of 0.3 from 2024 onwards, implying that if fishing pressure remains the predominant influence on the LFI, then the target is still achievable. As noted by Fung *et al.* (2012), the rate of recovery of the LFI has indeed slowed since around 2004, but this slowing is also evident in the modelled trend. The key point is that the modelled trend predicts a marked increase in the rate of recovery from around 2012/2013 onwards. The data should already be available to determine whether this pick up in the recovery rate has indeed occurred.

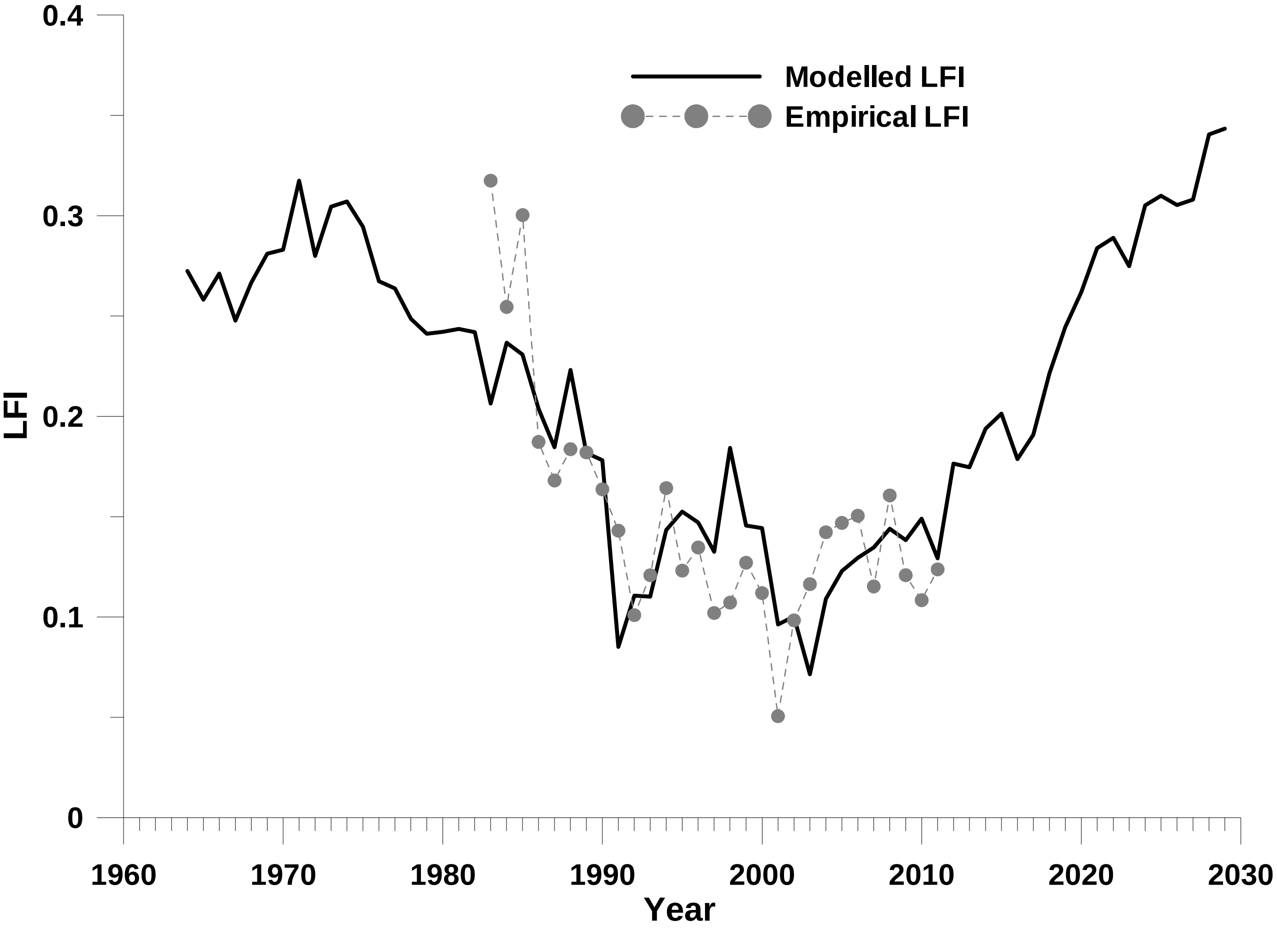

Figure 11. Modelled (see text) and empirical trends in the North Sea LFI.

Demersal fish species richness in the North Sea has increased in recent years and this has been linked to rising sea temperatures associated with climate change (Hiddink and ter Hofstede, 2008; ter Hofstede *et al.*, 2010; Simpson *et al.*, 2011). Much of this increase can be attributed to the influx of species, on increase in the population abundance, of species with a southern Lusitanian biogeographic affinity (Beare *et al.*, 2004). In the English Channel such environmentally driven increases in species abundance have been shown to be primarily restricted to small bodied species (Genner *et al.*, 2010). If a similar situation has occurred in the North Sea, the increase in species richness is primarily the result of environmentally driven increased small fish abundance, then this holds implications regarding the current target of 0.3 for the LFI.

Demersal fish species richness in the North Sea has increased with the Q1 IBTS suggesting an increase of 0.76 species per year over the full time series 1983 to 2011, but the rate of increase was particularly fast between 1988 and 2008 when species richness increased at a rate 0.95 species per year (Figure 12a.). Determining which species primarily drive the apparent increase in species richness is not straightforward; one would imagine that since there are around 70 species at the start of the time series and around 90 at the end, it is simply a case of identifying the 20 species that make up the difference. However, although species richness may have peaked at 90 species recorded in any one year, in reality 132 species were sampled during the full 29 years of data analysed. Most species are extremely rare and as a consequence are not sampled in each annual survey even when present in the surveyed area. Whilst some immigration of new species has occurred (see above), in reality most of the apparent increase in species richness was almost certainly driven by an increase in the abundance of species always present, such that the probability of their being sampled in the survey increased.

Figure 12b shows the frequency distribution of numbers of species encountered in a specified number of years. Figure 12c demonstrates that the species sampled in every year of the survey played no part in driving the positive species richness trend. Similarly the role of species sampled in between 22y and 28y of the survey was also minimal, but the 72 species sampled in 21y or fewer showed an increasing probability of being encountered later on in the time series and so contributed most to the observed increase in species richness. Of these 72 species, 25 had a Boreal affinity and 46 had either a Lusitanian or Atlantic affinity (1 species was a miss-identification, occurring only in the Caribbean), and of the Boreal species, 8 (32%) could attain an ultimate body length >40cm, while 17 (37%) of the Lusitanian/Atlantic species could grow to over 40cm. These 72 species generally contributed less than 1% to total North Sea demersal fish so their potential to really influence the LFI was minimal (Figure 12d). Interesting their proportional contribution to the biomass of ‘large’ demersal fish was larger, particularly from 2006 onwards (Figure 12d), so if anything their influence would be to promote recovery of the LFI, rather than to inhibit it.

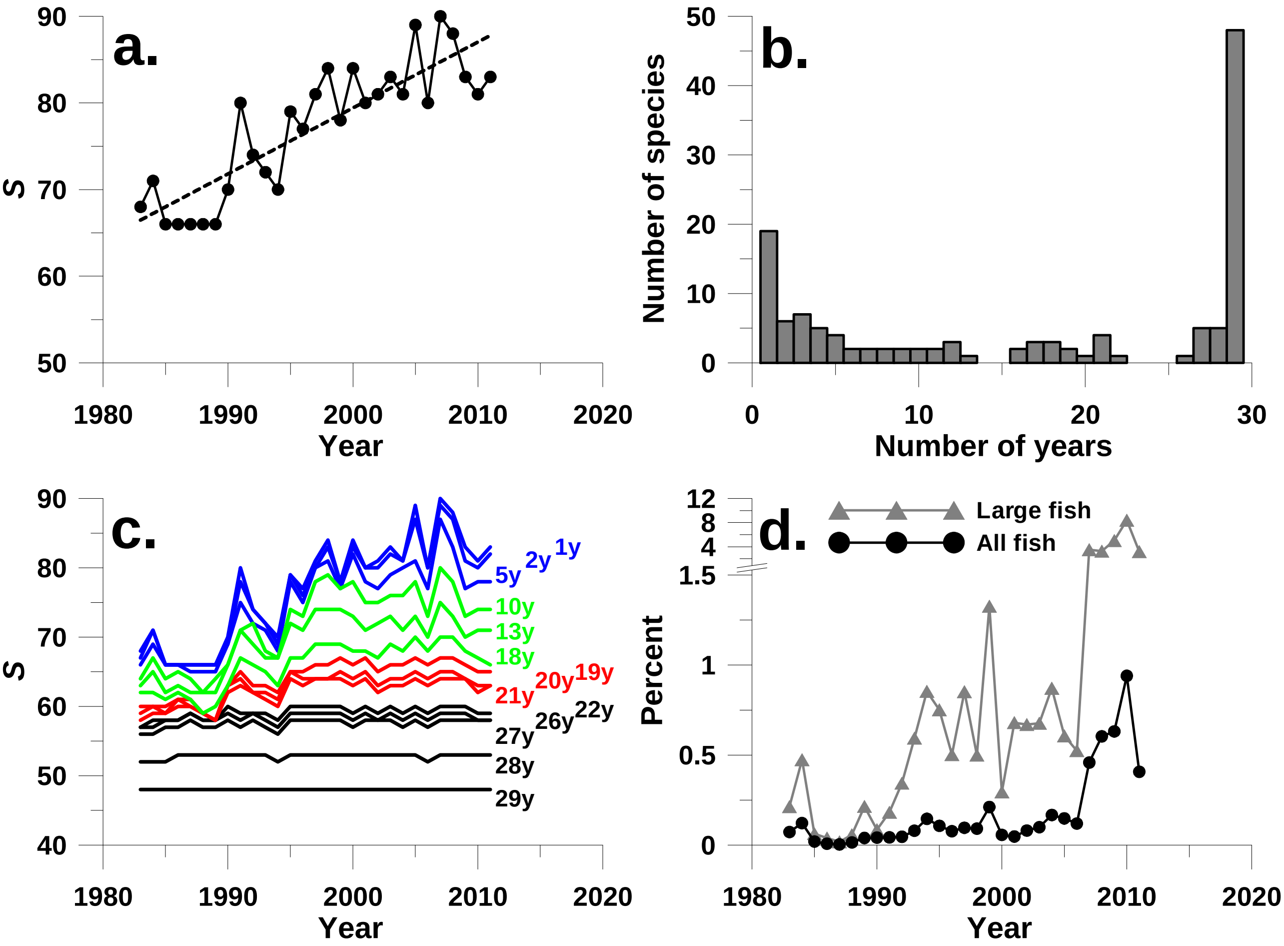

Figure 12. a.) Variation in the number of demersal fish species recorded in the Q1 IBTS each year. b.) Frequency histogram showing the number of species recorded a given number of years during the 29y time span of the Q1 IBTS. c.) Variation in annual demersal fish species count in the Q1 IBTS for species recorded a specified number of years or less. Thus the line labelled 29y includes only those species recorded every year, the right-most column in (b.), the line labelled 28y includes only those species recorded every year, the two right-most columns in (b.) and the line labelled 1y includes all species in (b.) and so duplicates the species richness plot in (a.). d.) Variation in the percentage contribution of the species driving the increase in species richness (species recorded in ≤21 y of the survey) to the biomass of all demersal fish and to the biomass of large (>40cm) demersal fish. Note the break in the y axis.

Figure 13 shows temporal trends in the biomass of both ‘large’ (>40cm) and ‘small’ (≤40cm) fish within the North Sea demersal fish community, reprising Figure 5 in Greenstreet *et al.* (2011), but in this instance the contributions of Boreal and Lusitanian/Atlantic species is also shown. The influence of variation in the biomass of large fish on the LFI remains clear, but what is perhaps unexpected is that proportion of this biomass contributed by species with a Lusitanian/Atlantic affinity has shown an increasing trend throughout the time series, but with a marked spike since 2006. Conversely, among the ‘small’ fish component, the proportion of biomass contributed by Lusitanian/Atlantic species, whilst varying considerably, has shown little in the way of any systematic trend. A pronounced peak in Lusitanian/Atlantic species biomass is evident around the late

1980s and early 1990s, and this is coincides with both the initial major decline in large fish biomass and the initial first major decrease in the LFI. These analyses suggest that the LFI target of 0.3 may in fact be remarkably robust to changes in the populations size of individual species coincidental changes in community species composition that have been linked to climate change (see above); indeed environment-related changes in the fish community may have had a greater influence on the initial decline in the LFI that previously suspected.

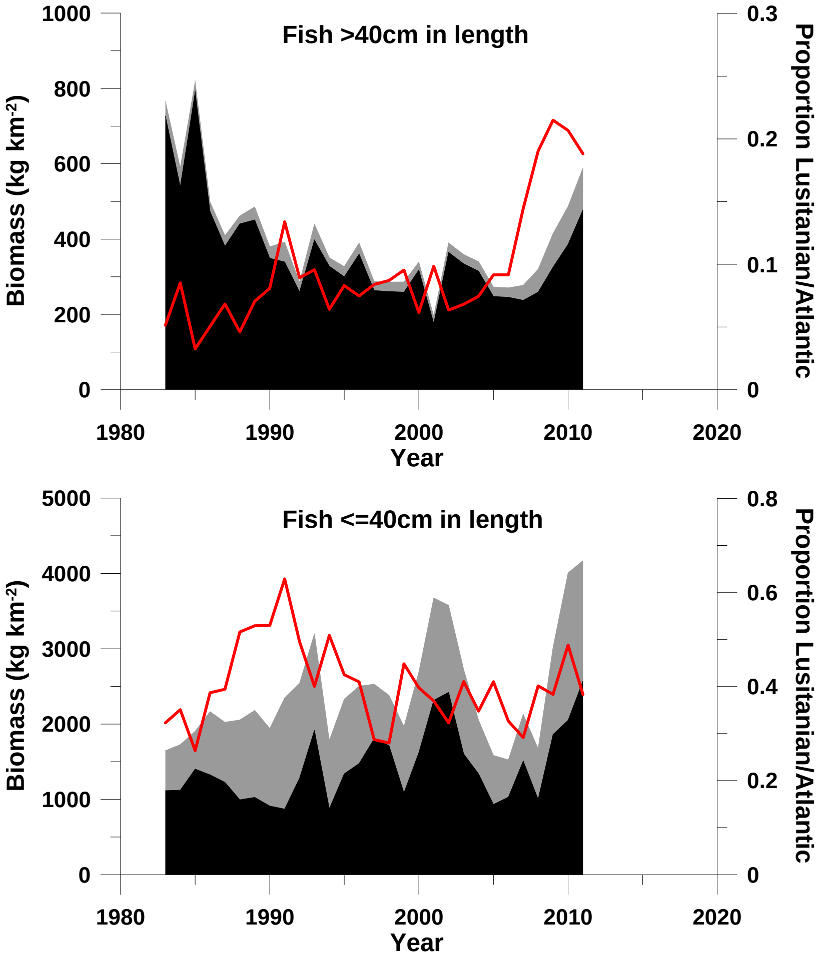

Figure 13. Trends in the mean biomass density of 'large' (>40cm) and 'small' (≤40cm) fish in the North Sea demersal fish community with Boreal (black fill) or Lusitanian/Atlantic (grey fill) biogeographic origin. Red line shows the proportion of the biomass consisting of fish of Lusitanian/Atlantic biogeographic origin.

In the North Sea variation in the biomass of cod and saithe exceeding a body-length of 40cm generally accounted for 60% of more of the total biomass of 'large' fish (Greenstreet *et al.*, 2011), and the well documented decline in the cod stock in particular accounts for a large fraction of the fall in the LFI through the 1980s. The future recovery of the LFI seems likely to depend heavily on future cod and saithe biomass trajectories. Both species are Boreal, and population size changes and distributions shifts have been documented, particularly for cod, that might suggest that the future prognosis for both species in the

North Sea might not be too optimistic. Landings of both species have declined markedly in recent decades as fisheries managers have attempted to reduce rates of fishing mortality, first to levels compatible with the precautionary approach (Rice, 2009), and now more recently to levels consistent with maximum sustainable yield (Figure 14). However, what is of particular concern is that recruitment in both species has shown a marked decline since the early 1980s, and currently medium term average recruitment is an unprecedented low level, and well below the long-term average level (Figure 15). Unless stock assessments take adequate account of these low levels of recruitment, then advice regarding acceptable catch rate levels could be over-optimistic leading to excessive fishing mortality that could inhibit recovery of the spawning stock biomass of these two species and potentially jeopardise recovery in the LFI.

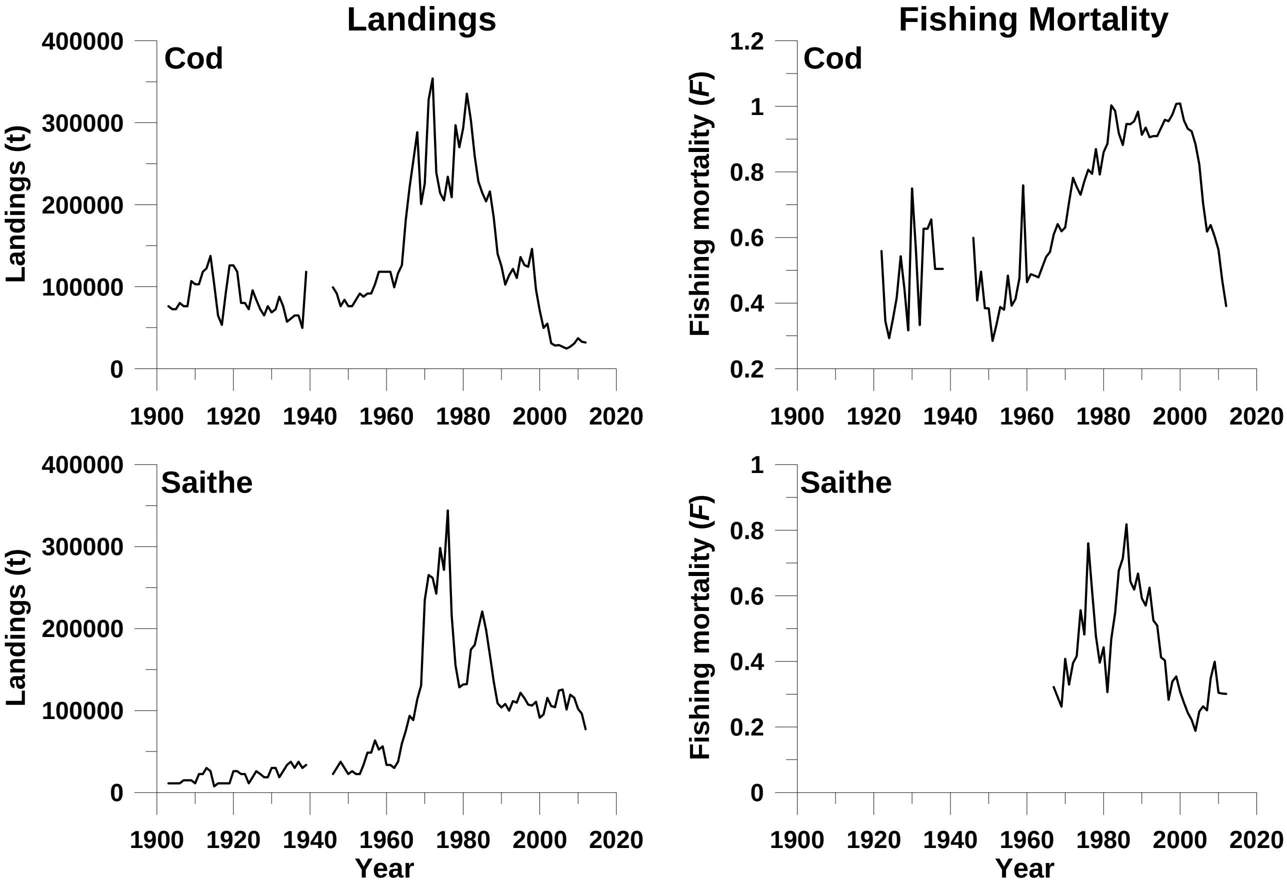

Figure 14. Trends in the landings and rate of fishing mortality of cod and saithe in the North Sea.

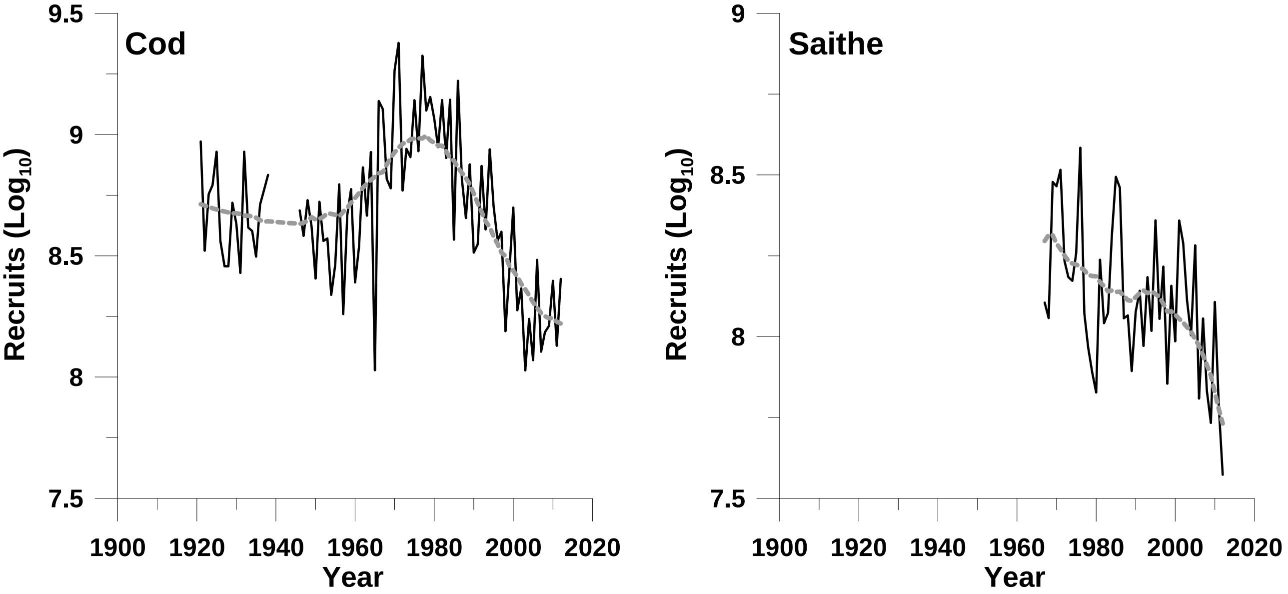

Figure 15. Trends in the rates of recruitment on (Log10 transformed). Dotted grey lines show fitted Loess smoothers.

Seabirds

Population abundance and breeding success of seabirds (B-1, B-2, B-3 and FW-1)

Three separate common indicators related to seabird breeding success have been proposed. B-2, the 'annual breeding success of kittiwakes', and B-3, the 'breeding success/failure of marine birds', are Descriptor 1 indicators supporting the requirement to maintain biological diversity, while FW-1, the 'reproductive success of marine birds in relation to food availability', is a Descriptor 4 indicator addressing the need to maintain food web structure and function (OSPAR, 2013b). A further common indicator for seabirds under Descriptor 1, B-1, addresses 'species specific trends in the relative abundance of non-breeding and breeding marine bird species' (OSPAR, 2013b). Reproductive productivity is a key variable influencing seabird population dynamics (Crespin *et al.*, 2006; Sandvik *et al.*, 2012). We therefore consider the effects of changing environemental conditions on the targets set for all four indicators together, since the same arguments are relevant to each indicator.

Interestingly the target for B-2, kittiwake breeding success, has the effect of environmental change built into the target definition. The target setting procedure is based on the relationship observed by Frederiksen *et al.* (2004) between kittiwake breeding success and sea temperature in the spring of the preceeding year (Figure 16). By monitoring sea surface temperature appropriately, kittiwake breeding success in any one year can be determined from this relationship and set as the target. Figure 16 shows the relationship obtained in years when no fishery for sandeel was in operation, and this is the relationship used for target setting. The figure also shows the impact of the fishery, causing a depression in the breeding success rate away from this target level.

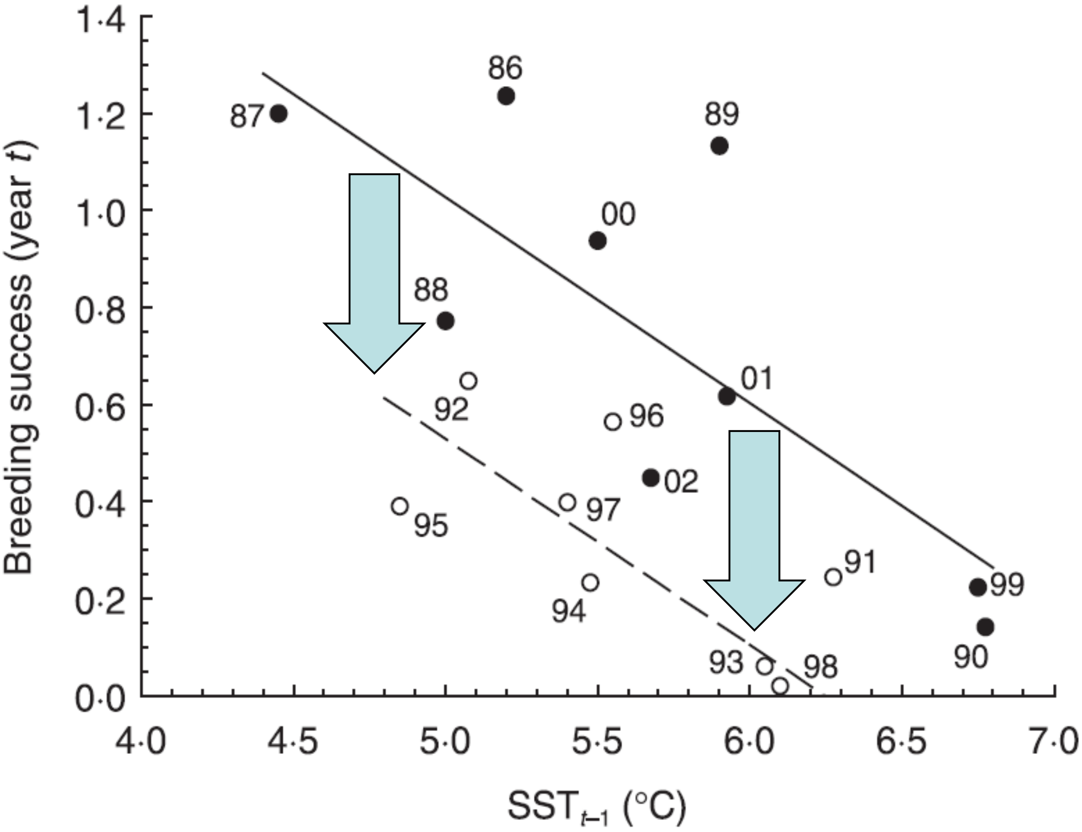

Figure 16 The relationship between kittiwake breeding success at the Isle of May, Firth of Forth, Scotland, and sea temperature during the preceding winter in years when no sandeel fishery was in operation on the nearby Wee Bankie (solid dots and line) and in years when the sandeel fishery was active (open dots and dashed line). Arrows indicate the negative displacement of the relationship in years when the fishery was active, indicating the reduction in sandeel availability to foraging adult kittiwakes caused by the fishery in these years. Modified from Frederiksen *et al.* (2004).

Indicators B-3 (breeding success/failure of marine birds) and FW-1 (reproductive success of marine birds in relation to food availability) were intended to compliment B-2, to cover other seabird species and to provide a similar breeding success indicator in areas or subregions where kittiwakes were not present (OSPAR, 2013a). FW-1 is a food web structure and function indicator because seabird breeding success has so frequently been linked to changes in the availability of prey (Cairns 1987; Harris and Wanless, 1997; Monaghan *et al.*, 1989; Frederiksen *et al.*, 2006; Daunt *et al.*, 2008) that it has often been proposed as an indicator of lower trophic level food web conditions (Cairns, 1987; Furness and Camphuysen, 1997; Frederiksen *et al.*, 2007; Iverson *et al.*, 2007; Piatt *et al.*, 2007a; Piatt *et al.*, 2007b; Parsons *et al.*, 2008). For both Indicator B-3 and Indicator FW-1, the target proposed at the Criterion level is "*widespread seabird colony failures (defined as the production of ≤ 0.1 chicks fledged per nest* (Cook *et al.*, 2012) *should occur rarely in other species that are sensitive to changes in food availability*". This Criterion target would be assessed on the basis of the number of species achieving species-specific supporting targets: "*The annual percentage of colonies experiencing breeding failure does not exceed the mean percentage of colonies failing over the preceding 15 years* (appropriate for species, such as the arctic tern *Sterna* *paradisaea*, that naturally frequently fail to breed), *or 5%* (appropriate for species, such as guillemots *Uria aalge*, that naturally rarely fail to breed), *whichever value is greater, in more than three years out of six*" (OSPAR, 2013b).

The evidence reviewed above suggests that changes in the availability of suitable fish prey to breeding seabirds are highly likely to occur as a result of projected warming of European seas associated with current climate change scenarios. Not only are small pelagic fish, such as sandeels, herring and sprats likely to be directly affected through environmental effects on their recruitment and physiology, but their zooplankton food supplies are also likely to be affected as the environment warms. Possible reductions in food availability linked to climate change may render the target of less than 5% of colonies experiencing breeding failures almost impossible to achieve. It has been recognised that using the mean value over the last 15y of the time series as the baseline for setting targets runs the risk of having a shifting baseline, and this risk is particularly high if the data display a trend. Climate change scenarios suggest a progressive warming sea temperature, and under these circumstances, the Frederiksen *et al.*, (2004) relationship (Figure 16) suggests that breeding success could progressively decline. It is likely that breeding failures will become increasingly more frequent, a declining trend in the time series, which could invalidate this target setting process.

A further common indicator under Descriptor 1 for seabirds, B-1, addresses 'species specific trends in the relative abundance of non-breeding and breeding marine bird species' (OSPAR, 2013b). The Criterion level target for this indicator is "*Changes in abundance of marine birds should be within individual target levels in 75% of species monitored*" (OSPAR 2013a). Current monitoring programmes provide data to estimate the abundance of 13 species of breeding seabirds in the Celtic Seas subregion and 16 species in the Greater North Sea (ICES, 2011). For each species-specific indicator of relative abundance, the supporting target is "*species-specific annual breeding abundance should be more than 80% of the baseline for species that lay one egg, or more than 70% of the baseline for species that lay more than one egg*" (OSPAR, 2013a). The setting of the baseline level is therefore of critical importance and three options have been suggested:

i. A point in the past when, based on expert judgement, anthropogenic impacts are likely to have been relatively minimal compared to the rest of the time-series; the baseline needs to reflect prevailing climatic conditions. It may prove difficult to set a baseline that meets both criteria.

ii. The mean value of the time series. This method carries the risk of a shifting baseline e.g. if a population is in long-term decline, the baseline will also decline as time goes on - so much so that target may eventually be met, without the population recovering.

iii. Where no previous data are available: set baseline at the start of the new time-series and amend in due course - see (a) and (b) (OSPAR, 2013a).

All three target setting methods are vulnerable to climate change impacts on breeding success. If reproductive productivity declines below species-specific minimum thresholds, then population abundance simply cannot be maintained. Historic population sizes will be unachievable, unless the climate change correction factor can be adequately factored in. Long-term mean values only provide shifting baselines, which will render their use as a basis for target setting extremely problematic. Even using the most recent abundance estimates will provide target values that are still too high and unachievable, if breeding failure becomes common place as a result of climate change.

Assessments of seabird status using these indicators will require careful interpretation. Changes in the marine environment arising from climate change could well cause a high percentage of 'failing GES' assessment outcomes. Investigation will be necessary to determine whether such outcomes really are due to inadequate management of human activities, or whether the targets are now inappropriate because of the changing environment.

Population abundance and pup production of marine mammals (M-3, M-4 and M-5)

Mirroring the approach to indicator development and target setting for seabirds, three similar reproductive productivity and population abundance indicators have been proposed for marine mammals:

i. Harbour seal and Grey seal pup production (M-5)
ii. Abundance of grey and harbour seal at haul-out sites & within breeding colonies (M-3)
iii. Abundance at the relevant temporal scale of cetacean species regularly present (M-4)

Just as with seabirds, reproductive productivety is intrinsically linked to population abundance so all three indicators and their targets are again considered together. For all three indicators, the preferred approach to target setting would involve the use of historical data to provide a baseline level representative of an acceptable situation. However, the lack of appropriate data is recognised and furthermore, even if the necessary data were available for the two seal indicators, its suitability for target setting is also questioned given environmental changes (e.g. coastal developments, erosion, etc.) that may have occurred during the intervening period. Instead, for both seal indicators, a type of trends based target has been proposed: for pup production this is "*No statistically significant long-term average decline of ≥10% at each Management Unit*", while for population abundance the proposed target is "*Maintain populations in a healthy state, with no decrease in population size with regard to the baseline (beyond natural variability) and restore populations, where deteriorated due to anthropogenic influences, to a healthy state*". Common indicator M-4 relates to four of the most common shelf-waters species, minke whale *Balaenoptera acutorostrata*, harbour porpoise *Phocoena phocoena*, bottlenose dolphin *Tursiops truncatus* and white-beaked dolphin *Lagenorhynchus albirostris*. Again the proposed target focuses on maintaining the current situation, "*Maintain populations in a healthy state, with no decrease in population size with regard to the baseline (beyond natural variability) and restore populations, where deteriorated due to anthropogenic influences, to a healthy state*". These trends based targets are unusual in that rather than aspiring to positive (recovery) trends, they set out to avoid negative (declining) trends, and as such implicitly assume that all three indicators already meet GES; the management objective is therefore to maintain GES and avoid deterioration in status.

Just as seabird reproductive productivity and population abundance may be affected by climate change related alterations to their prey resources, the same is true of marine mammals. Again therefore, failure to meet targets should be treated cautiously and merit further investigation to ensure that anthropogenic pressure really is to blame, rather than erroneous target setting that failed to take climate change into account adequately, before introducing more stringent management measures. However, many marine mammals are not restricted just to feeding at the small pelagic forage fish trophic level; they can also prey on larger sized fish and by prey-switch, could be more resilient to climated change related effects on their population dynamics.

Food webs

Reproductive success of marine birds in relation to food availability (FW-1)

This has been addressed under the birds indicators section

Size composition in fish communities (LFI) (FW-3)

This has been addressed under the fish indicators section

Changes in average trophic level of marine predators (cf MTI) (FW-4)

The Marine Trophic Index (MTI) was developed to assess the impacts of fishing on food webs and was adopted by the Conference of the Parties to the Convention on Biological Diversity (CBD). The MTI is calculated using estimates of species abundances and their mean Trophic Level (TL). Trophic level values range from the lowest, 1, for primary producers to the highest level, 5, for apex predators. The MTI adopted by the CBD included only species with mean TL > 3.25 in order to represent predators in the system: the indicator proposed for food webs by OSPAR (FW-4) is identical. The mean TL of a group should be estimated from local gut contents, isotope analyses or food web models (e.g. Ecopath with Ecosim). Alternatively, when none of the previous information is available, online data portals such as FishBase can be used. Abundances are best determined from scientific surveys, but the indicator has also been computed based on fisheries catch statistics (landings). If a food web model is available, time series of the TL indicators should be derived from the model for comparison to the survey data since the model can facilitate the inclusion of non-sampled or poorly sampled species in assessments. Model derived indicators incorporate food web interactions and thus develop our understanding of the impact of these interactions on the trophic level indicator.

OSPAR propose that the indicator should be within an "acceptable deviation from a baseline" and this target is made specific to an ecosystem through the determination of the appropriate baseline. The baseline for the MTI can be defined from a past state of the ecosystem in which the ecosystem was lightly impacted by fishing. If long time-series of survey data are unavailable OSPAR suggest that indicators based on catch data can be considered in order to determine a baseline level.

Shannon *et al.* (2014) undertook an extensive evaluation of TL indicators by making use of survey data as well as catch data and model-derived indicators in each of 9 well-studied marine ecosystems: Northern Adriatic Sea, South Catalan Sea, North Sea, West Coast of Scotland, Inner Ionian Sea, Guinean EEZ, Northern Humboldt, Western Scotian Shelf, and the Southern Benguela (Figure 17). Using detailed regional information and data on exploitation history, fishing intensity, and environmental conditions, TL indicators were evaluated in terms of their ability to capture fishing effects at the community level of marine ecosystems and reflect environmental change in chlorophyll a (Chl-a) and sea surface temperature (SST) (Table 6). The sign of the relationship between Chl-a and the survey-based TL indicator (i.e. the MTI with species of mean TL > 3.25) displayed great variability across regions. However for SST, 8/9 relationships with TL were negative and significant correlations were displayed in the North Sea and the area of the Western Coast of Scotland. In 7 of the 9 case studies, SST increased significantly with time suggesting a typically negative impact of warming on the trophic level of marine communities. If all surveyed species were included in the TL indicator (i.e. not only those species with TL > 3.25) the correlation coefficient between TL and SST increased and was significant in 5 of the 9 ecosystems (negative in North Sea, West Coast of Scotland, South Catalan Sea, Southern Benguela and positive in Inner Ionian Sea). So as SST increased over time, the mean trophic level of the community decreased due to increases in dominance of lower TL species in the survey data.

Despite general global agreement ($R = 0.65$, $p < 0.05$, $n = 9$) between the survey based TL indicator (with only groups with TL > 3.25) and the same indicator based on landings (only species TL > 3.25) the survey-data indicators often differed from the catch-based indicators in a specific ecosystem (only 4 / 9 ecosystems displayed individual significance). The authors found that both catch-based and survey-based indicators had to be considered to disentangle changes in community structure: catch based pressure indicators responded specifically to fishing, while survey based and model derived indicators also integrated environmental and food web effects on lower trophic levels (typically lightly fished or unfished). The trajectory of TL indicators was also found to be dependent on the particular exploitation history and fisheries management strategies adopted in an ecosystem and this context, along with change in the environment, is vital for interpreting change in such indicators.

Ideally, during target setting survey, model and catch based TL indicators should be used to identify the baseline and to identify the period in which fishing pressure was not overly altering the trophic structure of the system. Changes in the indicator due to the environment and climate over the time series of data availability and forecasts should be explored in order to determine a target robust to these drivers. Future trajectories of the TL indicators are likely to decrease as a result of further warming due to an increase in the abundance of low TL species.

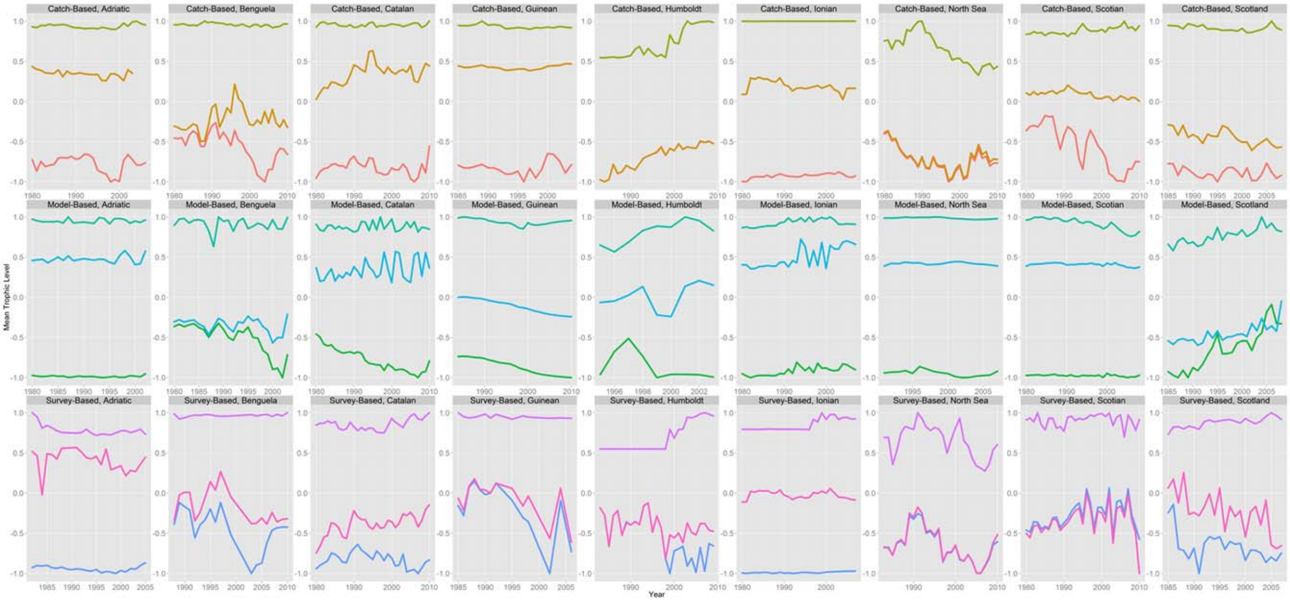

Figure 17: Trophic Level indicators determined from catch data (top), model output (middle) and survey data (bottom) for nine ecosystems. In each pane three lines are shown for indicators determined with all available species, only those with TL>3.25 or those TL > 4 (see legend). Figure adapted from Shannon *et al.* (2014).

Table 6. Correlations between TL indicators and fishing and environmental drivers (nd = no data, adapted from Shannon *et al.* 2014).

Sea area	Fishing effort	Fishing mortality	Catch/Biomass	Chl-a	SST
N Adriatic Sea	−0.42	**−0.44**	**−0.49**	0.50	-0.40
S Catalan Sea	0.27	**0.37**	**−0.69**	0.17	0.31
North Sea	**0.86**	**0.91**	**0.39**	0.10	**-0.48**
W Coast Scotland	nd	0.37	**0.57**	**0.90**	**-0.56**
Inner Ionian Sea	0.17	0.05	**−0.57**	0.70	-0.04
Guinean EEZ	−0.07	−0.08	−0.10	**-1.00**	-0.22
Northern Humboldt	−0.15	0.20	−0.09	-0.16	-0.22
W Scotian Shelf	nd	0.22	0.08	-0.52	-0.04
S Benguela	**0.59**	−0.05	**0.54**	0.00	-0.32

Plankton life forms (FW-5)

The metrics used to fulfil the FW-5 indicator role are identical to that used under PH-1, so refer to text above.

Concluding comments

The AMO can alternately mask or enhance the increase in SST associated with AGW in the northern Atlantic (Edwards *et al.*, 2013). The marked increase in SST over recent decades documented in most seas across the region is not just the consequence of AGW, but due to the superimposition of a positive AMO anomaly on top of the long-term trend in SST linked to AGW. Assuming a 60y cycle, the current warm phase of the AMO should end sometime around 2025, and seas across the north Atlantic should once again start to cool (Edwards *et al.*, 2013). However, given the increasing trend in SST associated with AGW, it seems unlikely that SST will drop below the average value observed over the 20th century, and so will remain comparatively warm. Nevertheless, with the negative residuals from this trend associated with a cool AMO phase, SST should at least drop once again to levels recorded towards the end of the 20th century, and so we should once more be able to draw on historic experience to set empirical targets for ecological indicators and estimate parameter values for ecosystem models.

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4 The Marine Strategy Framework Directive: a legal framework to support the application of integrated ecosystem assessment to implement ecosystem-based management across European seas

In this chapter, the following Terms of reference are addressed:

1. For the suite of indicators and targets proposed for Descriptor 1 ("Biological diversity is maintained) by Member States in support of the Marine Strategy Framework Directive, WGBIODIV will:
    - b. Consider the potential consequences of the introduction of management measures intended to achieve Good Environmental Status targets for particular ecosystem components on the capacity to achieve indicators/GES targets set for other ecosystem components.
    - d. Evaluate how metrics and indicators for various facets of marine biodiversity can be best integrated to derive more regional and holistic assessments of 'biodiversity status'.
4. Request from SIBAS: Identify, define and test activity-pressure-state links of indicators now and in future, including considering single/cumulative/synergistic effects of pressures.

Summary

The European Union (EU) Marine Strategy Framework Directive (MSFD) requires an "ecosystem-based approach to the management" of marine natural resources be implemented across European waters, but does not define what is meant by this term. Here the scientific and policy-related literature addressing the development and implementation of an "ecosystem-based approach" is reviewed to identify a precise terminology. Development of an ecosystem approach to management (EAM) has been an evolutionary process. The term EAM encompasses this whole process, which consists of four distinct phases, giving rise to the terminology adopted in this section:

- i. *Classical Fisheries Management* (CFM) which considers separate fish stocks in isolation of each other using simple population dynamics models (e.g. single species virtual population analysis (VPA)) utilising basic parameters for each individual stock. Only the aspirations of a single sector (fisheries) are considered.
- ii. An *Ecosystem Approach to Fisheries* (EAF) takes more account of broader ecosystem processes (e.g factors affecting recruitment) and of interactions between the

different targeted stocks (e.g. food web processes) that affect each stocks' dynamics. More complex models (e.g. multi-species VPA) are used, which can support management strategy evaluation (MSE), and so improve the scientific basis underpinning fisheries management decisions. Still only the aspirations of a single sector (fisheries) are considered.

iii. *Ecosystem-Based Fisheries Management* (EBFM) still only addresses the aspiration of the single (fisheries) sector, but more complex models, and relationships incorporating variables associated with other ecosystem components, are employed to estimate the impact of fishing activity on the broader marine ecosystem. This widens the perspective of MSE, allowing fisheries aspirations to be balanced against wider ecosystem consequences.

iv. *Ecosystem-Based Management* (EBM) takes EBFM to the next level by including the aspirations of multiple sectors (e.g. fishing, gravel extraction, shipping, renewable energy, etc.). At this level, MSE can address the needs of multiple sectors and impacts of each sector's activity on the marine ecosystem to maximise and balance the exploitation of marine natural resources and ensure sustainable use.

The MSFD addresses the needs of all sectors exploiting the full range of marine natural resources, but in such a way that the ecological consequences are not excessive or irreversible. The intention being that exploitation of all resources should be maintained at or just below levels that can be sustained over the long-term. The MSFD therefore requires EBM.

With each incremental phase in the development of the EAM, the level of integration required increases, but the precise shape of the integration required has been the subject of considerable debate. The literature contributing to this debate is reviewed. The conclusion that emerges is that for EBM to be made operational, a formal mechanism for processing the required integration is necessary. Just as individual stock assessments have traditionally provided the principle scientific basis supporting CFM, now a formal approach to integrated ecosystem assessments (IEA) is required to provide the scientific basis to support the EBM needed by the MSFD.

The need for, and development of, IEA has also been the focus of considerable scientific endeavour in recent years; again this literature is reviewed. A six step framework for IEA has emerged. Progress in implementing the MSFD to date is related to this framework to identify which aspects of an IEA have been completed, and to clearly establish what still needs to be done. This suggests that we are at a relatively advanced stage at scoping the IEA necessary to implement the MSFD and that good progress has been made with regard to the development of indicators and their targets, setting out the assessment process with respect to each indicator and with ensuring that the necessary monitoring programmes are in place to provide the data required to derive the indicators and carry out an assessment. However, to date little in the way of formal risk assessment has been carried out and the format of any formal management strategy evaluation is still largely undefined. The process for integrating the outcomes of individual indicator assessments has still to be decided.

Several different methods for integrating/aggregating the information conveyed from multiple assessments of a number of different individual indicators have been proposed. WGBIODIV examined some of the pros and cons associated with each method. A full

IEA to meet MSFD requirements involves integration of individual assessment outcomes at several different levels:

i. from an assessment of the status of an ecosystem component based on individual indicators;
ii. to integrating these indicator assessments to derive an assessment of the status of an ecosystem component at the Criterion level;
iii. then to integrating these Criterion level assessments to produce an assessment of the status of an ecosystem component at the Descriptor level;
iv. then to integrating the these Descriptor level assessments of the status of each ecosystem component across all ecosystem components to derive an overall assessment of state at the Descriptor level,
v. and finally, potentially having to integrate these Descriptor level assessments to determine overall status of the marine ecosystem.

Two simulation exercises confirm the fact that choice of integration method affects final IEA outcomes. Deciding which integration method is most appropriate for each situation, at each level of integration, is critically important and has the potential to influence the overall outcome profoundly. It is essential therefore that these decisions are taken *a priori* of actually carrying out the assessment.

Two further areas of difficulty when undertaking IEA to support the MSFD were identified. Firstly, selection of indicators to support EBM has to date primarily been done on a case by case basis, often using a set of selection criteria to inform selection decisions. The selection criteria invariably include the need for established pressure-state relationships, so that observed changes in state can be interpreted in such a way that specific advice as to how to manage pressure can be formulated. However, these pressure-state relationships have in the most part been considered in isolation. EBM addresses the need to manage multiple activities, leading to multiple pressures, so as to achieve preconceived goals for state. Thus IEA will need to address the cumulative impacts of multiple pressures on both single state indicators and multiple state indicators in such a way that MSE can subsequently identify the appropriate 'activity mix' that gives the best overall compromise between exploitation of all marine natural resources commensurate with acceptable, sustainable levels of deterioration in ecosystem status.

Secondly, just as the pressure-state relationships have generally been considered only on an indicator by indicators basis, the approach to target setting has been similar. Targets have been set for each indicator using a variety of different baselines and reference points, with different logic often underpinning each approach. To date little consideration has been given to assessing whether the targets set for seabird indicators, for example, are compatible with targets being set for fisheries management, or for fish communities. WGBIODIV examined the targets being set for different indicators for different ecosystem components and identified several potential inconsistencies. The key 'take-home' message from this analysis is that should an indicator fail to meet its target, one should first question whether the target is actually appropriate, before immediately assuming that the management measures put in place are inadequate and placing further restrictions on the human activities involved.

Introduction

The Marine Strategy Framework Directive (MSFD) represents the legal framework by which Member States (MSs) of the European Union (EU) will implement an "ecosystem-based approach to the management" of marine natural resources across European seas. However, precisely what is meant by an ecosystem-based approach to management, and exactly what this would entail, is never explained within the MSFD document. Developing an ecosystem approach to manage marine natural resources has received considerable attention in the scientific literature. Many different terms have been used and a variety of different definitions have been suggested. Frequently the same or very similar terms have been defined differently by different authors, and often different terms have been assigned similar definitions (Arkema *et al.*, 2006; Belgrano and Fowler, 2011). This has generated confusion as to exactly what the different terms mean (Hirshfield, 2005; Arkema *et al.*, 2006), and as a consequence it is not completely clear what management action is needed to fulfil MSs' obligations under the MSFD to adopt an ecosystem-based approach to management.

At the policy level, the concept of an ecosystem approach to manage human activities was first developed by the Convention of Biological Diversity (CBD) and principles were established in connection with the fifth Conference of the Parties, Decision V/6 (https://www.cbd.int/decision/cop/default.shtml?id=7148), covering ecological, socio-economic and governance aspects. The concept and principles were adapted to fisheries management by UN Food and Agriculture Organization (FAO), which resulted in the Ecosystem Approach to Fisheries (EAF) endorsed by FAO's Committee on Fisheries in 2003 (FAO, 2003).

As the concept of ecosystem-based approach to management has developed, the need for integration at many levels has become increasingly apparent (Sissenwine and Murawski, 2004); integration of the different strands of scientific knowledge on which scientific advice to managers is based; integration of policy objectives for different sectors; integration of the planning and management procedures used to regulate the different sectors; and finally integration of the ecosystem impacts associated with each sector (Misund and Skjoldal, 2005; Rosenberg and McLeod, 2005; Rice, 2005). Integrated ecosystem assessment (IEA) is therefore an integral component of an operational ecosystem approach to management (EAM) (Link and Browman, 2014), IEA will therefore have a key role to play in successful implementation of the MSFD. However, IEA has been defined in different ways and in different contexts and as a consequence it is often not completely clear what IEA entails in practice and how it should be carried out (Link and Browman, 2014). This can be partly explained by the fact that the EAM is itself ill-defined; different practitioners use similar terminology applied to different types of management with different management goals. As IEAs become more complex, and the level of integration involved increases, additional problems emerge that require to be overcome, such as addressing the cumulative impacts on marine ecosystem components associated with multiple pressures linked to a range of different anthropogenic activities. Having to deal with potential inconsistencies between targets set for individual indicators also becomes more of an issue as the numbers of indicators and associated targets increases.

In this chapter therefore we first review literature relating to the development of an ecosystem approach to the management of marine natural resources. Our aim is to determine what is meant by an EAM, and what implementation of an EAM involves. We then examine the MSFD itself to establish the level of EAM necessary to satisfy MSs' obligations under the MSFD. Having achieved this we consider the various definitions of, and approaches to, IEA to establish the type of IEA required to support successful implementation of the EAM across European seas and so meet the goals of the MSFD. We then review the progress to date towards actually undertaking the type of IEA that will be required to assess the state of the marine ecosystems in waters that come under the jurisdiction of the MSFD. Finally we start to scope some of the emerging difficulties in order to start the process of finding solutions.

The need for an ecosystem approach to management

Towards the end of the 20th century and into the early 21st century, evidence of global scale over-exploitation of fish stocks was increasingly apparent (Jackson *et al.*, 2001; FAO, 2002, Myers and Worm, 2003; Hilborn *et al.*, 2003: Worm *et al.*, 2006; Worm *et al.*, 2009), leading to the assertion that the traditional single-sector, single-species approach epitomised by classical fish management had failed (Schiermeier, 2002; Piet and Rice, 2004; Lotze, 2004; Mace, 2004; Caddy and Seijo, 2005; Schwach *et al.*, 2007). Awareness of the broader impact of fisheries on components of marine ecosystems beyond just the commercial species targeted by fisheries was also increasing (Watling and Norse, 1998; Jennings and Kaiser 1998; Hall, 1999; Gislason and Sinclair, 2000; Kaiser and de Groot, 2000; Chuenpagdee *et al.*, 2003; Sinclair and Valdemarsson, 2003; Frederikson *et al.*, 2004), along with concern over the potential threat that this posed to other goods and services provided by marine ecosystems that might be of value to society (Pauly *et al.*, 1998; Pauly *et al.*, 2000; Balmford *et al.*, 2002; Worm *et al.*, 2006). Consequently, by the turn of the century calls for a change in management approach were wide-spread, with an increasing majority advocating an ecosystem approach to management, or ecosystem-based management (Garcia *et al.*, 2003; Sissenwine and Mace, 2003; Sainsbury and Sumaila, 2003; Cury, 2004; Jennings, 2004; Pikitch *et al.*, 2004).

The political response has been swift and global (Agardy 2005). The United Nations Convention on Biological Diversity offered a definition and principles for the implementation of an ecosystem approach to management stressing the "*equitable promotion of conservation and sustainable use*" (CBD 2004). The Reyjavik Declaration of 2001 required nations to apply an ecosystem approach with respect to the exploitation of marine natural resources (Cury, 2004). This was reinforced at the World Summit of Sustainable Development in Johannesburg in 2002, eventually leading to the introduction of legislation to enforce this change in attitude. For example, in the USA, the 2006 amendment and reauthorization of the Magnuson-Stevens Fishery Conservation and Management Act, known as the Sustainable Fisheries Act, provided the basis for altering the way that fisheries were managed to ensure greater protection for the wider marine environment (Fluharty, 2005). Likewise in Europe, introduction of the MSFD in 2008 provided the legal framework by which Member States (MSs) of the European Union (EU) would implement an ecosystem approach to management of marine natural resources (EC, 2008; 2010). This intent is explicitly stated throughout the MSFD (EC, 2008), for example:

- Paragraph 8 of the preamble states "*By applying an ecosystem-based approach to the management of human activities …* ";

- Then further on, paragraph 44 states "*Programmes of measures and subsequent action by Member States should be based on an ecosystem-based approach to the management of human activities …*";
- Then paragraph 3 of Article 1 states "*Marine strategies shall apply an ecosystem-based approach to the management of human activities, ensuring that the collective pressure of such activities is kept within levels compatible with the achievement of good environmental status and that the capacity of marine ecosystems to respond to human-induced changes is not compromised, while enabling the sustainable use of marine goods and services by present and future generations*".

However, while Article 3 of the MSFD defines many of the terms used in the document, presumably to promote clarity and facilitate a common understanding by all the MSs bound by the MSFD, nowhere does the MSFD document actually define what is meant by an "*ecosystem-based approach*", or set out exactly what such management entails.

Establishing a consistent terminology

Arkema *et al.* (2006) use the term 'ecosystem-based management' and define this to include the management of species, other natural commodities, and humans as part of the ecosystem using adaptive, ecosystem-level planning that encompasses cross-jurisdictional management goals, and which utilises long-term monitoring programmes to track the impact of human activities and the effectiveness of management measures. But they also point out that too many different terms have been used, with far too many different definitions associated with them, often with multiple definitions applied to the same term, or the same term used to mean different things. These terms include 'ecosystem management' (Grumbine, 1994; Larkin, 1996; Stanford and Pool, 1996; Brussard *et al.*, 1998; Yaffee, 1999), 'ecosystem approach to management' (Beattie, 1996; Griffiths and Kimball, 1996; Murawski, 2007), 'ecosystem approach to fisheries' (Garcia *et al.*, 2003; Sissenwine and Mace, 2003; Sissenwine and Murawski, 2004; Cury, 2004, Jennings, 2004), 'ecosystem-based fisheries management' (Pikitch *et al.*, 2004; Babcock and Pikitch, 2004; Hilborn *et al.*, 2004), 'ecosystem-based sea use management (Douvere, 2008), 'ecosystem-based management' (Guerry, 2005; Arkema *et al.*, 2006), to list but a few. In one single book, four separate terms are used ('ecosystem-based management', 'ecosystem approach to fisheries management', 'ecosystem-based management of fisheries', 'ecosystem approach to fisheries'), all in a similar context (Fanning *et al.*, 2011).

After reviewing the literature, Arkema *et al.* (2006) identified 17 separate aspects of 'ecosystem-based management' and applied a statistical analytical approach to determine whether the various definitions for the different terms actually differed – did the different terms mean different things? They concluded that there was no difference in the way that the different terms were being used and that essentially they all meant approximately the same thing. They suggested therefore that the proliferation of terminology and variety of definitions only served to promote confusion; scientists should settle on one universal term and over-arching definition in order to stimulate coherent progress towards development of the science necessary to support the transition to ecosystem-based management. Walther and Möllman (2014) agree, believing that lack of a clear definition of the ecosystem approach to management causes problems in communicating to stakeholders what this involves and is therefore a major impediment to its implementation.

Link and Browman (2014) describe a process by the EAM has developed: from the single-stock perspective of early fisheries management to the need to account for and address the detrimental impacts of all human activities across the entire marine ecosystem. They define four distinct levels at which ecosystem based management might be adopted, assigning a specific term to each level:

1) "Classical Fisheries Management" (CFM) lies at one end of the spectrum. CFM focuses on the commercial species targeted by fisheries using stock-assessment models that incorporate a few essential processes, e.g. age-based rates of both natural and fishing mortality, growth, and annual recruit cohort abundance, with parameter values that are non-dynamic and relate only to the focal species (Beverton and Holt, 1957; Ricker, 1975; Rice 2011). Whilst proponents accept the simplicity of CFM, they also recognise that single species population dynamics are influenced by trophic and competitive interactions with other species in the marine ecosystem.

2) An "Ecosystem Approach to Fisheries" (EAF) is the next level in Link's and Browman's (2014) scheme. The EAF utilises multi-species models that explicitly incorporate the trophic and competitive interactions either ignored or parameterised using constants in traditional stock assessment models that support CFM to provide more dynamic estimates of natural mortality, growth and recruit abundance by relating these terms directly to variation in predator, prey and competitor abundance (Sissenwine and Daan, 1991; Sparre, 1991; Field *et al.*, 2006; Garrison *et al.*, 2010). Multi-species models allow exploration of costs and benefits of fishing different species at different levels, so facilitating management strategy evaluation. Link and Browman (2014) describe this approach to management as an EAF because its management goals still only address the needs of this single sector; 'using what is known about the ecosystem to manage fisheries' better; it is therefore essentially an extension of CFM (Fluharty, 2005). However, Link's and Browman's use of the term EAM in this sense is perhaps the most contentious since this term, perhaps more than any other, has been associated with other meanings in both the scientific and policy-related literature. The section below detailing the FAO perspective provides a prime example of this.

3) The next level, "Ecosystem Based Fisheries Management" (EBFM) takes the EAF a stage further by adding further predator (e.g. marine mammals and seabirds), prey (e.g. benthic invertebrates and zooplankton) and habitat (e.g. seabed integrity) components to the multi-species models. This markedly increases the scope of management strategy evaluation possible (e.g Latour *et al.*, 2003; Hall *et al.*, 2006; Pope *et al.*, 2006; Andersen and Pedersen, 2009; Speirs *et al.*, 2010; Heath, 2012; Heath *et al.*, 2014), promoting even better management of fisheries, but also enabling the impacts of fisheries on these wider ecosystem components to be evaluated as well. Management strategy evaluation can now include the need to trade-off social and economic goals for the fishing industry against requirements to conserve other ecosystem components. Link and Browman (2014) call this EBFM, not full EBM, because it still only focuses on a single sector; the ecological impact of fishing. It does not address the needs of

other sectors, or their impact on the ecosystem, and it does not facilitate examination of trade-offs between fisheries and other stakeholders' interests.

4) Full "Ecosystem Based Management" (EBM) constitutes the fourth and final level. EBM takes account of all sectors; the goods and services that can be provided by the ecosystem are properly ascertained, individual and cumulative impacts of all sectors on marine ecosystems are fully evaluated, and cross-sector trade-offs to identify the mix of goods and services that can best benefit society over the long term are assessed. At a strategic level, EBM considers both ecological and human objectives in the exploitation of natural resources (Curtin and Prellezo, 2010), addresses the various natural and anthropogenic pressures faced by key components of ecosystems (Link, 2002), and aims to maintain ecosystems in a healthy, productive, resilient condition whilst still providing key marine resources for human consumption (Arkema *et al.*, 2006; Rosenberg and Sandifer, 2009; Berkes, 2012). In summary, EBM places "emphasis on a management regime that maintains the health of the ecosystem alongside appropriate human use of the marine environment, for the benefit of current and future generations" (Jennings, 2004).

Rice (2011) also considers four distinct components that constitute an ecosystem approach to fisheries management:

1) Improving traditional single-species orientated fisheries management by taking better account of environmental and biological drivers that affect the population dynamics of harvested stocks.
2) Minimising the detrimental footprint of fishing on the wider marine ecosystem by taking account of the impact of fishing on other ecosystem components and managing fisheries accordingly.
3) Broadening the governance of marine ecosystems to include the needs and impacts of all stakeholders operating in the marine environment.
4) The introduction of integrated management as a means of balancing the needs and impacts of all uses of marine natural resources and ensuring that the aggregated impacts from all anthropogenic activities do not exceed sustainable limits.

The first three of these components exactly describe the three step-wise processes involved in progressing between the four levels defined by Link and Browman (2014). Link's and Browman's (2014) paper introduced a themed collection of papers that addressed the issue of Integrated Ecosystem Assessment (IEA). The concept of IEA portrayed in these papers exactly chimes with the fourth component of the ecosystem approach to fisheries management described by Rice (2011). However, while Link's and Browman's (2014) and Rice's (2011) views on what constitutes the ecosystem approach to fisheries management are remarkably similar, their use of terms differs.

Rice (2011) points out that the scientific community uses the terms EAM, EAF and EBM interchangeably; a point we also make above and one borne out by the results of Arkema's *et al.* (2006) analysis. Rice (2011) prefers the term EAM to EBM, considering that the former implies an incremental process that extends conventional fisheries management principles to include wider ecosystem considerations, while the term EBM is more appropriate to defining specific management scenarios. However, in his paper Rice

(2011) only uses the term EAM once, in making this distinction between the EAM and EBM. Instead, Rice (2011) uses the term ecosystem approach to fisheries (EAF), clearly meant in the sense of an ecosystem approach to fisheries management. In this sense, Rice uses the two terms EAF and EAM synonymously, but makes a distinction between these terms and EBM. A key difference between the two papers is that whereas Link and Browman (2014) apply a separate term to each of the four levels at which an ecosystem approach to management might be applied, Rice (2011) uses only a single term applied to the whole process: all four components combined. Rice considers the term EBM best describes specific management scenarios. In a sense, Link and Browman (2014) agree since they only apply this term to their fourth management approach level. Rice (2011) believes that the term EAM best describes the whole incremental progression through all four levels, whereas Link and Browman (2014) provide no collective term that covers all four of their levels. Thus between the two papers, a definitive terminology emerges that we can adopt for this report: henceforth termed the Rice/Link and Browman terminology.

Following Rice (2011), we now use the term EAM to refer to the whole four-stage incremental process by which EBM has evolved from CFM, and we use the term EBM to refer to the end point of this evolutionary process; the fourth and last level described by Link and Browman (2014) (Figure 1). Walther and Möllman (2014) provide possibly the most concise definition of EBM (as we now use this term), as *"to manage natural resources in a holistic way, by considering the interacting influences of multiple use sectors on the environment … … to assure the health of the ecosystem alongside appropriate use of the environment for the benefit of future generations"* (although in fact they applied this definition to an EAM). This chimes exactly with Murawski's (2007) understanding of the two terms. Like Rice (2011), Murawski (2007) considers an EAM to mean the "incremental process" by which CFM has evolved into EBM; defining an EAM as "extending existing management foci (e.g. fisheries) to include additional considerations consistent with ecosystem management characteristics", and EBM as "a management scheme primarily designed to address overall ecosystem considerations". Following Link and Browman (2014), we now use the terms EAF and EBFM to refer to the second and third levels of the EAM developmental process described by Link and Browman (2014). Thus EAF refers to improved CFM that takes greater account of environmental and biological processes affecting target species population dynamics; EAF therefore considers the single sector and takes negligible account of the impact of fishing on the broader ecosystem (Figure 1). EBFM refers to management that, whilst still only considering the activities of the single sector, fisheries, nevertheless seeks to manage these activities in such a way as to mitigate the detrimental impact of fishing on other components of the marine ecosystem (Figure 1).

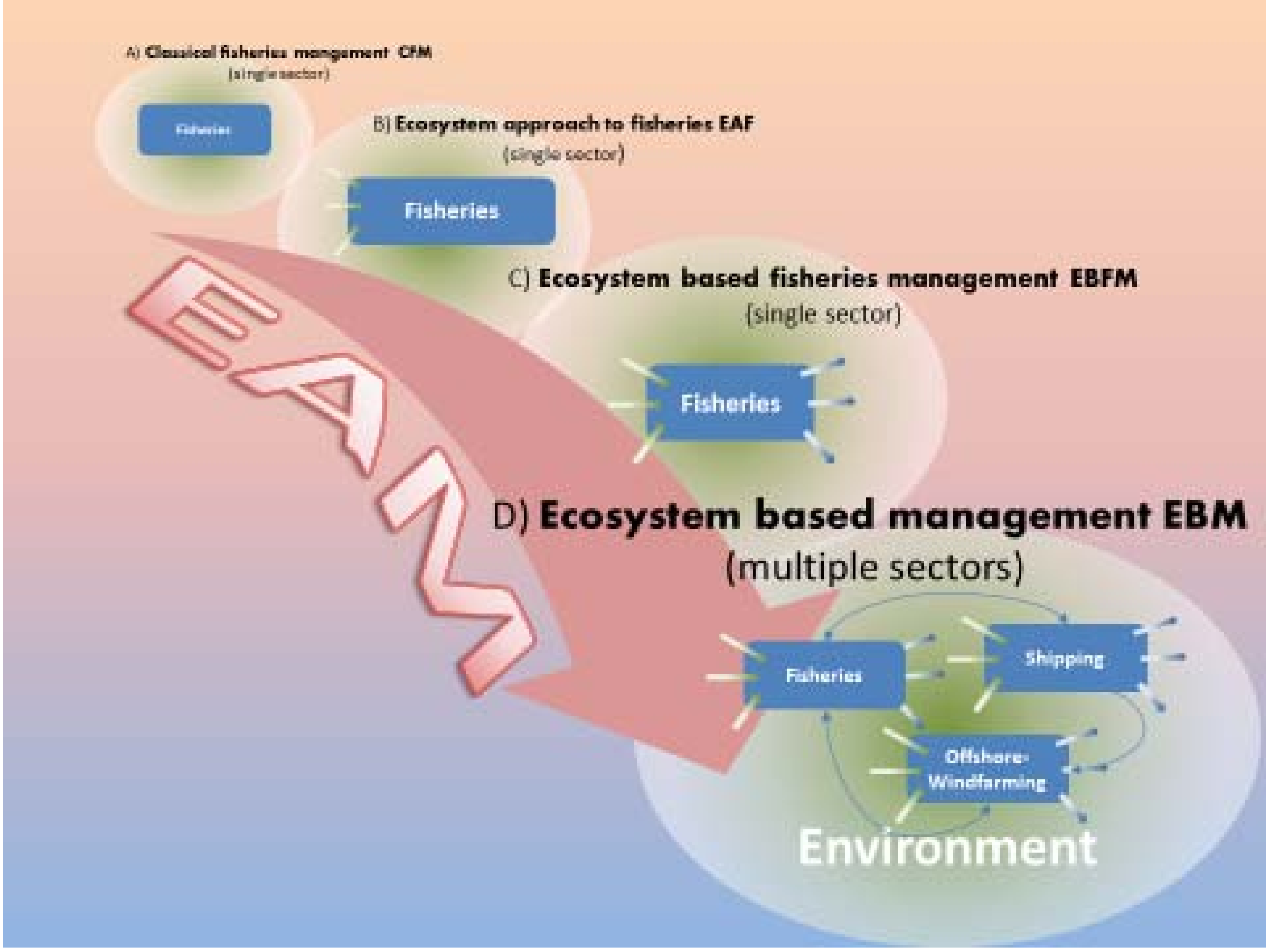


Figure 1. Illustration of different levels in the concept of an 'ecosystem approach to management' (EAM) and resulting Rice/Link and Browman terminology derived from Rice (2011) and Link and Browman (2014). The term EAM applies to the whole incremental development process that starts with 'classical fisheries management' (CFM), progresses through the 'ecosystem approach to fisheries' (EAF), then 'ecosystem based fisheries management' (EBFM), and culminates in 'ecosystem based management' (EBM): each level represented by a 'bubble'. Green inward arrows represent greater utilisation of ecological and environmental information in managing the human activity in question to achieve social and economic objectives. Blue outward arrows represent awareness of the impact of the activity on the broader marine ecosystem resulting in a management process that balances the need to maximise social and economic objectives for the activity against requirement to minimise its detrimental impact on the wider ecosystem and environment. Blue boxes in the bubbles represent different human activity sectors. See text for further details.

The Rice/Link and Browman terminology illustrated in Figure 1 potentially provides the clarity needed (e.g. Arkema *et al.*, 2006) to move forward towards implementing an EAM at some level. However, before we can fully adopt this terminology, we need to ensure that it can be reconciled to all previous uses of these, and other, terms in the literature.

Reconciling the Rice/Link and Browman terminology to previous usage

As discussed above, Rice (2011) uses the terms EAM and EAF synonymously to refer to the whole incremental process by which the EAM has developed. We suggest the term EAM be applied to the entire process because Link and Browman (2014) also use the term EAF, but apply it to just one specific level of the EAM (Figure 1). Rice uses the term EAF primarily because this is the term adopted by the FAO. So it is now germane to question exactly what the FAO means by the term EAF. Rice (2011) cites FAO (2003), which provides a set of guidelines to supplement the FAO 'Code of Conduct for Responsible Fisheries' (CCRF). In this section we first examine the FAO's perspective of the EAF, both from the text of the CCRF and from subsequent technical publications (FAO 2003; Garcia *et al.*, 2003). But as already alluded to, the EAM of marine resources has been the subject of numerous other scientific publications, and these have used a large number of different terms in a wide range of different contexts. We then review this literature to determine whether all these different terms, and all their different uses in various contexts, can be reconciled to the terminology emerging from Rice (2011) and Link and Browman (2014). If successful, then the Rice/Link and Browman terminology (Figure 1) will be adopted throughout the rest of the report.

The FAO Perspective

The Code of Conduct for Responsible Fisheries

The UN Food and Agriculture Organization (FAO) Code of Conduct for Responsible Fisheries (CCRF) (1995) represents "soft" international law (Scandol *et al.*, 2005); mainly voluntary guidelines that outline the principles and operational procedures for sustainable fisheries. However, parts of the CCRF are based on relevant rules of international law such as the United Nations Convention on the Law of the Sea. The FAO CCRF actually makes no mention of the EAM, EAF or any other similar term. Here we review the CCRF to see which parts are relevant to the EAM, and which levels of an EAM they have most relevance for.

Article 2 provides the overall objectives of the Code, paragraph (a) states to "*establish principles . . . for responsible fishing and fisheries activities, taking into account all their relevant biological, technological, economic, social, environmental and commercial aspects*", while paragraph (g) states "*promote protection of living aquatic resources and their environments and coastal areas*". Article 6 states some general principles of the Code. Paragraph 6.1 recognises the importance of aquatic ecosystems: "*States and users of living aquatic resources should conserve aquatic ecosystems*", and paragraphs 6.2 and 6.8 expand on this theme, respectively stating "*Management measures should not only ensure the conservation of target species but also of species belonging to the same ecosystem or associated with or dependent upon the target species*" and: "*All critical fisheries habitats in marine and fresh water ecosystems, such as wetlands, mangroves, reefs, lagoons, nursery and spawning areas, should be protected . . .* ". This requirement for fisheries to be managed in way that safeguards marine environments and ecosystems raises the EAM proposed in the FAO CCRF to at least the level of EBFM In Link's and Browman's (2014) terminology.

Article 7 of the CCRF provides specific guidance for the management of fisheries: considering in turn, general principles, management objectives, management frameworks and procedures, data gathering and management advice, the precautionary approach (e.g

Richards and Maguire, 1998; Hilborn *et al.*, 2001; Rice, 2009), management measures, implementation and financial institutions, all primarily directed towards ensuring that the yield from fisheries is maximised over the long-term: i.e. maximum sustainable yield. At best, this article mainly addresses the CFM and EAF levels of an EAM (Figure 1). However, paragraph 7.2.2.d, addressing management objectives, requires that management measures should ensure that "*biodiversity of aquatic habitats and ecosystems is conserved and endangered species are protected*", while paragraph 7.5.2, discussing the precautionary approach, requires that "*States should take into account … … the impact of fishing activities, including discards, on non-target and associated or dependent species*". It is only these two paragraphs in the section addressing management of fisheries that, following Link's and Browman's (2014) terminology, raise the EAM espoused within the FAO CCRF from that of an EAF to that of EBFM. Paragraph 7.2.2.f states "*adverse environmental impacts on the* (fisheries) *resources from human activities are assessed and, where appropriate, corrected*" and paragraph 7.2.2.g explicitly lists pollution as one of these activities. This is the first indication of the FAO CCRF moving beyond the EBFM level to the EBM level of the EAM. It requires mitigation of the negative effects on fisheries resources of human activity in other sectors, but it does not imply any need for consideration in the reverse direction; the need to mitigate the detrimental consequences of fishing on capacity to achieve other sectors' objectives. As such the FAO CCRF still cannot be considered to espouse full EBM.

Article 8 deals with fishing operations, in which paragraphs 8.7.1 to 8.7.4 address protection of the marine environment, but only considering the need to minimise pollution from vessels actively engaged in fishing operations. The remainder of the article specifically advises on best practices within the fishing industry. Only one paragraph moves beyond fishing and fisheries; paragraph 8.4.7 requires that "*States should ensure that assessments of the implications of habitat disturbance are carried out prior to the introduction on a commercial scale of new fishing gear, methods and operations to an area*". This requires consideration of the environmental impact of fishing prior to new fisheries starting up, but it places no requirement on States to mitigate the environmental impact of current fishing activities.

Article 10, which addresses integration of fisheries into coastal area management, is the principle instrument within the CCRF that relates to the EAM. In particular:

- Paragraph 10.1.1 requires States to take account of "*the fragility of coastal ecosystems and the finite nature of their natural resources and the needs of coastal communities*";
- Paragraph 10.1.2. states "*In view of the multiple uses of the coastal area, States should ensure that representatives of the fisheries sector and fishing communities are consulted in the decision-making processes and involved in other activities related to coastal area management planning and development*";
- Paragraph 10.1.4 requires that "*States should facilitate the adoption of fisheries practices that avoid conflict among fisheries resources users and between them and other users of the coastal area*";
- Paragraph 10.2.2 suggests that "*In order to assist decision-making on the allocation and use of coastal resources, States should promote the assessment of their respective value taking into account economic, social and cultural factors*";

- Paragraph 10.2.4 requires that "*States, in accordance with their capacities, should establish or promote the establishment of systems to monitor the coastal environment as part of the coastal management process using physical, chemical, biological, economic and social parameters*"
- Paragraph 10.2.5 requires that "*States should promote multidisciplinary research in support of coastal area management, in particular on its environmental, biological, economic, social, legal and institutional aspects*"

The first bullet point requires fisheries managers take account of the impact of fishing on the marine environment, whilst all remaining bullet points require that the interests of the other sectors operating in the marine environment are also taken into account. However, a key issue with respect to Article 10 is that, while it addresses integration of fisheries into coastal area management, at no point does it define what is meant by the coastal area. If by coastal area it literally means the coastal zone, e.g. the six mile limit of UK national waters, then Article 10 holds little relevance to the way that fisheries are managed in European waters for example. On the other hand if by coastal area the FAO means the continental shelf, then Article 10 would appear to be moving towards advocating EBM.

The Ecosystem Approach to Fisheries

Garcia *et al.* (2003) first address what is meant by the term 'ecosystem management'. This concept emerged around the mid-1960s (Czech 1996; Czeck and Krausman, 1997), but the 1972 Stockholm Conference on the Human Environment and 1992 Rio de Janeiro Conference on Environment and Development (UNCED) and Convention on Biological Diversity (CBD) raised its profile. The concept is mainly derived from terrestrial systems, related to the direct manipulation of populations and habitats, and human activity, in both space and time with the specific purpose of optimising the long-term returns to humans (Lackey, 1998; 1999). Lackey (1999) defines 'ecosystem management' as "*the application of ecological, economic, and social information, options, and constraints to achieve desired social benefits within a defined geographic area and over a specified period*" while Cortner *et al.* (1994) suggest an alternative but similar definition, "*a management philosophy which focuses on desired states rather than system outputs and which recognizes the need to protect or restore critical ecological components, functions and structures in order to sustain resources in perpetuity*". 'Ecosystem management' therefore aims at (Grumbine, 1994; Larkin, 1996):

1) maintaining viable populations of all native species *in situ*;
2) representing within protected areas all native ecosystem types across their natural range;
3) maintaining evolutionary and ecological processes;
4) managing over periods of time of sufficient duration to maintain evolutionary potential of species and ecosystems; and
5) accommodating human use and occupancy within these constraints.

Putting this into a fisheries context, the FAO concluded that 'ecosystem management' consists of "*management decisions which involve a broad awareness of the consequences of fishing or other human actions to an ecosystem … … the necessity of understanding multispecies interactions and questions of altered structure of the biological community (ecosystem stability)*" (FAO-ACMRR, 1979). As far as the FAO is concerned therefore, successful 'ecosystem management' in the marine environment must take account of the potentially damaging effects to ecosystem structure and function of all human activities, not just fishing.

The CBD refers simply to the 'ecosystem approach', defined as "*Ecosystem and natural habitats management… to meet human requirements to use natural resources, whilst maintaining the biological richness and ecological processes necessary to sustain the composition, structure and function of the habitats or ecosystems concerned*". But this term is usually used in the sense 'ecosystem approach to …' (e.g. EAM or EAF) (Garcia *et al.*, 2003). Such an approach "*recognizes explicitly the complexity of ecosystems and the interconnections among its component parts*" (Fisheries and Oceans Canada, 2002), and is generally assumed to require:

1) definition and scientific description of the ecosystem in terms of scale, extent, structure, functioning;
2) assessment of its state in terms of health or integrity as defined by what is acceptable to society;
3) assessment of threats; and
4) maintenance, protection, mitigation, rehabilitation, etc., using:
5) adaptive management strategies.

The FAO Technical Consultation on Ecosystem-based Fisheries management held in Reykjavik in September 2002 adopted the term EAF. Ward *et al.* (2002) define an EAF as "*an extension of conventional fisheries management recognizing more explicitly the interdependence between human well-being and ecosystem health and the need to maintain ecosystems productivity for present and future generations, e.g. conserving critical habitats, reducing pollution and degradation, minimizing waste, protecting endangered species*". The use of the phrase "*extension of conventional fisheries management*" in this quotation chimes well with the process of evolutionary development characteristic of the EAM that is implicit in the terminology derived from Rice (2011) and Link and Browman (2014), and illustrated in Figure 1. Garcia *et al.* (2003) point out that definitions for the words 'ecosystem', 'approach' and 'fisheries' are easily obtained from a dictionary, and stringing these definitions together implies a process that uses specific means to achieve specified objectives. Again as the EAM has developed, so it objectives have become better defined and measures required to achieve these objectives have become better understood.

The Reykjavik FAO expert consultation determined that "*The purpose of an ecosystem approach to fisheries, … … is to plan, develop and manage fisheries in a manner that addresses the multiple needs and desires of societies, without jeopardizing the options for future generations to benefit from the full range of goods and services provided by marine ecosystem*". Consequently, the definition of the EAF associated with this purpose was "*An ecosystem approach to fisheries strives to balance diverse societal objectives, by taking into account the knowledge and uncertainties about biotic, abiotic and human components of ecosystems and their interactions and applying an integrated approach to fisheries within ecologically meaningful boundaries*". It is clear from this definition, and this stated purpose, that the FAO uses the term EAF to mean a governance system that takes account of multiple sectors operating in the marine environment, and the need to mitigate the detrimental consequences of these multiple human activities on the structure and functioning of marine ecosystems so as to ensure the long-term capacity of these ecosystems to provide the full range of ecosystem good and services desired by society. The FAO attach the same meaning to the term EAF that

Link and Browman (2014) associate with EBM; the final level in the evolutionary development of an EAM (Figure 1).

The guidelines to the implementation of the EAF developed by FAO in 2003, and endorsed by the Committee on Fisheries the same year, provide the rationale, the basic principles of ways of implementation. Further guidance was developed by FAO including, for example, the human dimension of EAF (FAO, 2009). During the past decade the FAO concept of the EAF, has been described in various documents (e.g. Bianchi and Skjoldal 2008; FAO, 2012a; 2012b). Recognizing the need of developing more practical guidance, FAO has a developed a toolbox (FAO, 2012). As part of the introduction, the toolbox provides, in very simple terms, what EAF entails.

The EAF is a sector based approach to sustainable fisheries and is primarily applied in the context of addressing issues that are caused by and/or can be controlled by the sector itself. However, the approach also helps identifying external factors impacting the sector that are beyond the control of fisheries and aquaculture authorities and stakeholders. Examples include coastal development, mining, pollution from land-based activities etc. If any of these factors are identified as undermining the sustainability of the resource base, links have to be developed with the competent authorities to find ways to mitigate these impacts and/or negotiate trade-offs. The key features of the framework proposed in the FAO guidelines for planning and implementing under the EAF can be summarized as follows:

1) a management plan is developed for a very specific area/system with operationally defined boundaries;
2) stakeholder participation is envisaged at all levels of the planning and implementation steps;
3) all key components of a fishery system (ecological, social-economic and governance), are comprehensively considered while also taking into account external drivers;
4) sustainability issues that need attention are identified and prioritized through a formal process (e.g. risk assessment);
5) management objectives related to environmental and social/economic aspects are reconciled including explicit consideration of trade-offs between them;
6) an adaptive management process is established that includes mechanisms for feed-back loops at different time scales to adjust the tactical and strategic performance based on past and present observations and experiences;
7) ‘best available knowledge’ is the basis for decision-making, including both scientific and traditional knowledge, while promoting risk assessment and management and the notion that decision making should take place also in cases where there is lack of detailed scientific knowledge;
8) the system builds on existing management institutions and practices.

Development of fisheries management plans is a key element in the implementation of these integrated approaches. It should be noted that the CCRF (FAO, 1995) also explicitly requires that “*Long-term management objectives should be translated into management actions, formulated as a fisheries management plan or other management framework*”. Implementation of an ecosystem approach requires, perhaps more explicitly than under conventional fisheries management and the CCRF, that management plans be developed by explicitly taking into account the three dimensions of sustainability. The planning process consists largely of examining existing or developing fisheries to identify key objectives (ecological, social and economic), priority issues to be addressed in order to move towards these objectives and the action required. The main result of this planning process is the backbone of EAF fisheries management plans. The EAF also prepares the fisheries agency to be part of multi-sectorial governance, e.g EBM).

The key steps for planning and implementing management plans integrated across sectors are very similar to what described above. The main differences can be summarized as follows:

1) The authority for developing and implementing the multisectoral management plan needs to be established, for example through legislation. In addition to facilitating the planning process, the authority will be responsible for monitoring that the plan is adequately implemented by all the participating sectors.
2) Setting sustainability objectives coherently across the different sectors. These have to be consistent with policy goals related to ecological, social and economic dimensions of sustainability as reflected in relevant policies.
3) Allocation of spatial and temporal distribution of human activities in the given area/ecosystems (including development of a use conflict compatibility map).

In adopting the term EAF, the FAO explicitly discarded the term EBFM (Garcia *et al.*, 2003). Garcia *et al.* (2003) cite the definition of EBFM proposed by the US National Research Council (1998): "*an approach that takes major ecosystem components and services — both structural and functional — into account in managing fisheries ... … It values habitat, embraces a multispecies perspective, and is committed to understanding ecosystem processes … … Its goal is to rebuild and sustain populations, species, biological communities and marine ecosystems at high levels of productivity and biological diversity so as not to jeopardize a wide range of goods and services from marine ecosystems while providing food, revenues and recreation for humans*". This term was not adopted at the 2001 FAO Reykjavik Conference perhaps because States inferred that is places emphasis on management of the economic activity and that the "ecosystem" might become the new 'foundation' of fisheries management, giving environmental considerations pre-eminence over socio-economic and cultural ones and raising concerns over socio-economic costs and feasibility. However, this definition talks of the 'goods and services from marine ecosystems' in the plural, places value on 'habitat' and stresses the need to rebuild 'biological communities and marine ecosystems'. It clearly relates not only to the impacts of fishing on marine ecosystems, but also the need to balance the needs and impacts of all sectors. As such the FAO's use of the term EBFM is close to the same sense applied to the term EBM by Link and Browman (2014).

Garcia *et al.* (2003) start their paper by stating that the meanings of the terms, such as "ecosystem management", EBM, EBFM and EAF, are not universally defined, and that these terms have been used to describe related concepts. This implies considerable overlap in their meaning (e.g. Arkema *et al.*, 2006), and indeed, the definitions given by Garcia *et al.* (2003) for the three terms, EAF, EBFM, and EAM, suggest only minimal and subtle differences in their meaning at best. The FAO may have adopted the term EAF, but still appears to use all three terms in a way that is synonymous with the term EBM as used by

Link and Browman (2014); all three terms relate to the need to manage all human activities in the marine environment in such a way as to maintain the capacity of marine ecosystems to provide the full range of ecosystem good and services required of society. So whilst the FAO's definitions of these terms might differ from the terminology derived from Rice (2011) and Link and Browman (2014), the FAO's use of these terms can still be reconciled to this new terminology.

Terms used in the scientific literature

Review of literature

Jennings (2004) uses the term EAF and maintains that an EAF is a significant step towards sustainable use of the marine environment, believing that the broad purpose of the EAF is to plan, develop and manage fisheries in a manner that addresses the multiple needs and desires of societies, without jeopardising options for the future. This could imply that the principle aim of an EAF, in Jennings (2004) view, is to mitigate the detrimental impacts of fishing on other components of marine ecosystems, suggesting a use of the term EAF that is synonymous with Link's and Browman's (2014) use of the term EBFM. However, while the bulk of Jennings' (2004) text focuses on the fisheries sector, the paper is also quite clear that meeting the needs of this sector must be balanced against the "multiple needs and desires of societies … … to benefit from the full range of goods and services (including of course non-fisheries benefits) provided by marine ecosystems"; Jennings (2004) therefore actually uses EAF in a sense that is synonymous with the use of the term EBM by Link and Browman (2014).

Jennings (2004) and Rice (2011) both use the term EAF, and both appear to use it in the same sense; to refer to a management process that takes account of the needs and impacts of multiple users of marine natural resources, balances these needs and impacts, and addresses cross-sector trade-offs. This is commensurate with Link's and Browman's (2014) use of the term EBM. Further, Jennings (2004) states that an EAF is part of the ecosystem approach (to management). Here the term EAM, the term preferred by Rice (2011), is clearly being used as an overall collective phrase, encapsulating the incremental process by which EBM in the marine environment has evolved from CFM; exactly the reason why Rice preferred this term. Jennings (2004) and Rice (2011) use the terms EAF, EAM and EBM almost synonymously, and in a sense that seems to reflect the end point in the development of the EAM; the end of the spectrum characterised by Link and Browman (2014) as EBM. In doing so, both Jennings (2004) and Rice (2011) use the term EAF in a way that directly aligns with the use of this same term by the FAO.

Larkin (1996) uses the term marine ecosystem management; believing that the growing need for this "stems from concerns about overexploitation of world fisheries and the perceived need for broader perspectives in fisheries management". His main concerns focus on the need to manage fisheries so as to minimise their impact on the broader marine ecosystem and so safeguard the alternative goods and services that marine ecosystems may provide for society. Lotze (2004) also uses the term ecosystem management and again mostly addresses the need to minimise the detrimental impact of fisheries on the wider marine ecosystem so as to maintain its overall health and functioning. In this respect, both Larkin (1996) and Lotze (2004) consider ecosystem management in the same sense that Link and Browman (2014) use the term EBFM. However, Lotze (2004) also recognises that marine ecosystem management will also need to address the damage caused by other sectors associated with "habitat destruction, nutrient loading, pollution and other disturbances", bringing Lotze's view of ecosystem management closer to what Link and Browman (2014) term EBM. Fluharty (2005) considers an EAM, but explicitly relates this to the management of fisheries. Fluharty uses this term to refer to a style of management that not only uses "what is known about the ecosystem to manage fisheries" more productively, but which also identifies the detrimental effects of fishing on marine ecosystems so that these can be alleviated. In this sense, Fluharty (2005) also addresses what Link and Browman (2014) regard as EBFM.

Hirshfield (2005) uses the term ecosystem approach, although whether to management, e.g. an EAM, or to fisheries, e.g. an EAF, is never clarified. He also uses the terms EBM and EAF and this paper is perhaps a prime example of the confusion of terminology that concerned Arkema *et al.* (2006). Indeed Hirshfield (2005) himself points out that "we need ecosystem-based management, but we're still not sure what it means". His paper focuses mostly on the need for minimising the detrimental effects of fishing on non-target species and habitats, and therefore mostly addresses EBFM as Link and Browman (2014) define the term. However, Hirshfield (2005) does explicitly state that the use of ecosystem approaches "will require those who manage ocean users (in particular those who manage fishers) to give greater value to the condition of non-fish components of the ecosystem", at least implying than an ecosystem approach will need to address the aspirations and detrimental impact of other sectors over and above just the fishing industry. Later, Hirshfield (2005) more explicitly states "For conservationists, ecosystem-based management encompasses all of the threats to ocean ecosystems, including pollution and non-fisheries based habitat destruction". Thus Hirshfield (2005) appears to use the terms EAM, EAF and EBM synonymously, using all three terms in the same sense that Link and Browman (2014) apply to the term EBM.

In introducing the theme section "Perspectives on ecosystem-based approaches to the management of marine resources" published in Marine Ecology Progress Series (2004, 274: 269-306), Browman and Stergiou (2004a) "focus on the fisheries sector". Thus, while the theme section title uses yet another term, ecosystem-based approaches to management, in the context of the terminology established by Link and Browman (2014), Browman's and Stergiou's (2004a) paper actually only addresses EBFM. It considers the broader impacts of fishing on the marine ecosystem and the use of management measures, such as marine protected areas (MPAs), to mitigate these impacts on key species and habitats; it does not explicitly consider the needs and impacts of other sectors, or the type of management strategy evaluation that would be necessary to establish optimal cross-sector trade-off strategies. Rosenberg and McLeod (2005) also use the term ecosystem-based approach to management, but in this instance it is clear that the authors consider that such an approach is required in order to address "a proliferation in ocean-related activities", "growing demands for marine resources", "manage fisheries", "mitigate the effects of coastal development", "improve water quality", and "manage numerous other sectors". Rosenberg's and McLeod's (2005) use of the term ecosystem-based approach to management is thus synonymous with EBM in the terminology established by Link and Browman (2014). Sherman *et al.* (2005) also use the term ecosystem-based approach to management, as well as the term EAM, but in a way that suggests both are used synonymously. They use both terms in a context that takes account not only of the wider impacts of fishing on marine ecosystems, but also of the impacts of other anthropogenic activities. Like Rosenberg and McLeod (2005), Sherman's *et al.* (2005) paper therefore also addresses EBM, as Link and Browman (2014) define this term. Sumaila (2005) also uses the term ecosystem-based approach to management as well as the term EBM. Sumaila (2005) again seems to use both terms synonymously, and since he considers the exploitation of marine natural resources in its broadest sense, Sumaila (2005) also uses both terms in the sense that Link and Browman (2014) define EBM.

Cury (2004) asserts that a "multi-disciplinary approach is needed for the EAF" to "help managers recognise and understand ecological limits to avoid loss of ecosystem integrity and to maintain fisheries in viable states". At no point does Cury consider the impacts on the ecosystem of other sectors or the need for cross-sectorial trade-offs, so the paper does not address EBM as defined by Link and Browman (2014). Neither are the goals of the EAF, as Cury considers it, completely clear. If the loss of ecosystem integrity is as least as important as maintaining viable fisheries then this use of the term EAF is perhaps synonymous with the term EBFM as defined by Link and Browman (2014), but if the aim is primarily to improve fisheries management through incorporation of wider ecosystem processes, then Cury's use of the term EAF matches this use of the term by Link and Browman (2014). Sissenwine and Murawski (2004) also use the term EAF, and because their definition appears to "consider multiple external influences" and to "balance diverse societal objectives", their use of this term initially appears synonymous with the term EBM established by Link and Browman 2014. However, their paper again focuses exclusively on the need to improve the effectiveness of fisheries management and on the need for measures to minimise the impacts of fishing on other components of the marine ecosystem deemed important by sections of society, or which might provide goods and services of value to society. For the most part therefore, this paper also uses the term EAF in the same sense that Link and Browman (2014) have established for the term EBFM.

Sissenwine and Mace (2003), Roux and Shannon, 2004; Valdimarsson and Metzner (2005), and Watson-Wright (2005) also all use the term EAF. In each case, the author's principal focus lies with a single sector, the fishing industry: ensuring that this sector does not compromise the integrity of other aspects of marine ecosystems (Sissenwine and Mace, 2003); or on achieving "responsible fisheries management" that through an understanding of ecological processes, patterns and limits, produces human benefits that are fairly distributed, does not cause unacceptable change in marine ecosystems thereby avoiding the loss of ecosystem integrity, and maintains fisheries in viable and sustainable states (Valdimarsson and Metzner, 2005); or balances the demands by fisheries on forage fish species against the needs of marine top predators that compete with the fishery for this fish resource (Roux and Shannon (2004); or considers the need to manage fisheries so as to reduce the bycatch of non-target species, and takes into account the need to protect fragile features of the seafloor habitat from damage by fishing gears (Watson-Wright, 2005). Sissenwine and Mace (2003), Roux and Shannon (2004), Valdimarsson and Metzner (2005) and Watson-Wright (2005) again all use the term EAF in the sense that Link and Browman (2014) relate to EBFM.

Browman and Stergiou (2004a) also use the term EAF; for example, stating "Both EAF and MPAs implicitly recognise that the value (to humanity) of the whole ecosystem is much greater than the sum of its parts". In this respect, the use of the term EAF appears synonymous with the term EBFM established by Link and Browman (2014). In a second paper in the theme section, Browman and Stergiou (2004b) consider MPAs as a central element of EBM. At points in their paper they appear to use the term EBM when considering the general development of an EAM, but their main text focuses on the use of MPAs to mitigate the impact of fishing on the wider marine ecosystem; with respect to the terminology established by Link and Browman (2014), their paper once more essentially addresses EBFM. In a later paper, Browman and Stergiou (2005) broaden this earlier perspective. They continue to use the two terms EBM and EAM (they actually use this abbreviation to stand for ecosystem-*based* approach to management) apparently synonymously, but they now explicitly state that implementation of a holistic EAM will "need to address the cumulative impacts of all activities (extractive or otherwise) on the ecosystem". This is now fully in line with Link's and Browman's (2014) use of the term EBM.

Hilborn (2004) discusses the need for EBFM in order to prevent fisheries causing the collapse of marine ecosystems (Jackson *et al.*, 2001; Myers and Worm, 2003) and the destruction of marine habitat (Watling and Norse, 1998). Babcock and Pikitch (2004) also discuss the increased need for EBFM in order to comply with the increasing number of laws, treaties, and agreements that require nations to minimise the impact of over-exploitation by fisheries, not only on the individual commercial fish stocks themselves, but also on other non-targeted components of marine ecosystems. In their paper, Tudela and Short (2005) use the term EBFM in the title; in the text, however, they also use the terms EBM and EAF, appearing to use all three terms synonymously. Tudela and Short (2005) primarily focus on the need, if EBFM is to be successfully implemented, to define ecosystem-based reference levels for fisheries management based on indicators of overall ecosystem structure and functioning with established thresholds that define ecosystem over-fishing. Frid *et al.* (2005) also use the term EBFM, and in relation to the increasing need in the northeast Atlantic for fisheries to be managed in such a way as to reduce their detrimental effects on the wider ecosystem. Hall and Mainprize (2004) discuss the potential for expanding the single-species reference point basis for CFM to include non-target species and other ecological indicators. Similarly, Latour *et al.* (2003) compare different types of models to explore their potential to support management that optimises both fisheries and conservation objectives. Scandol *et al.* (2005) comment on the fact that in Australia, in moving towards EBFM, fisheries operators are now required to undertake environmental impacts assessments. Link (2002) considers EBFM to be necessary to improve the implementation of ecosystem considerations into fisheries management. In all eight examples, since the focus is firmly fixed on a single sector, the fishing industry, and on minimising the impact this sector has on the broader marine ecosystem; the term EBFM is clearly being used in the same sense that Link and Browman (2014) use the term, and therefore fits the terminology illustrated in Figure 1.

Zeller and Pauly (2004) use the term EBM, but their paper again only appears to consider a single sector; the need for improving fisheries management by adopting a more ecosystem-based perspective so as to alleviate the damaging consequences of over-exploitation on the overall state of marine ecosystems. Hughes *et al.* (2005) also use the term EBM, but again their paper primarily addresses the need to mitigate the detrimental impacts of fishing on the broader ecosystem. With respect to Link's and Browman's (2014) terminology, both papers therefore also address EBFM, not EBM.

Web of Science meta-analysis

To explore how the use of the terms applied by Link and Browman (2014) to define the different levels in the development of an EAM has varied over time, a search of the Web of Science literature data base was made on 19th and 20th March 2015. Searches were made on the three key terms, EAF, EBFM and EBM. All results for the terms EAF and EBFM related to management of marine ecosystems. For the key term EBM, results were filtered by including 'marine' as a second search-term. Figure 2 shows the number of papers published per year that used each term. For the last period, 2013 to 2015, the number of years was assumed to equal 2.2 since we are still only part way through 2015. From the late 1990s to around 2006, the number of scientific papers published using each of the three terms steadily increased, and all three terms were used with approximately equal frequency. However, while the numbers of papers published per year using the terms EAF and EBFM has continued to increase at approximately the same rate, the numbers of papers using the term EBM has increased at a markedly higher rate. Since 2010, the numbers of papers published each year using the terms EAF and EBFM may actually have stabilised at around 20 per year, whereas there seems to have been no such waning in scientific interest in EBM.

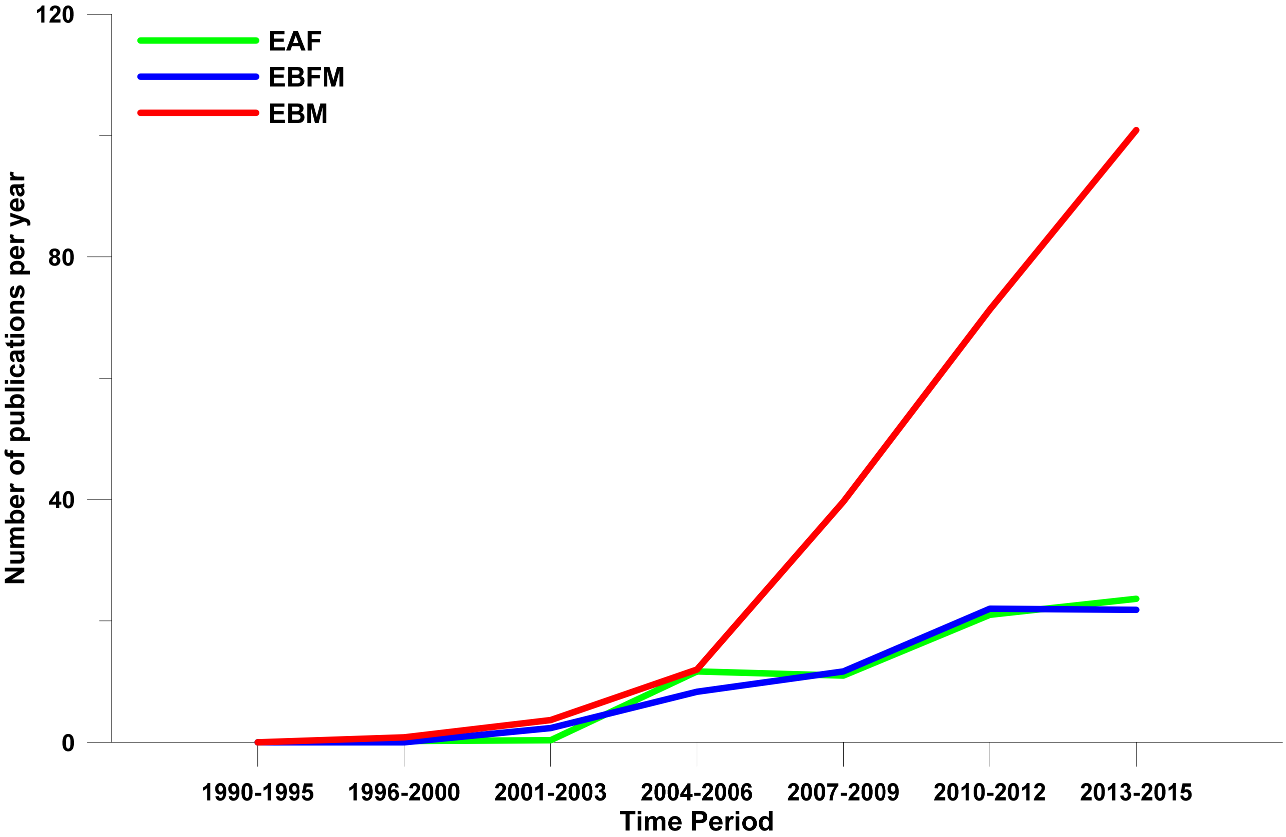


Figure 2. Number of scientific peer-reviewed papers published each year using the terms ecosystem approach to fisheries (EAF), ecosystem based fisheries management (EBFM) and ecosystem based management (EBM) (in the marine environment) in their title, abstract or key-words.

With the exception of Arkema's (2006) paper, which highlighted the confusion over the use of different EAM terms, and Rice's (2011) and Link's and Browman's (2014) papers that establish the terminology illustrated in Figure 1, papers reviewed so far to reconcile use of different terms to the Rice/Link and Browman terminology were all published before 2006. Given the continuing increase in the publication rate of papers addressing the EAM issue, and particularly the marked increase in the publication rate of papers using the term EBM since 2005 (Figure 2), a sample of 100 papers, all published from 2006 onwards, was reviewed to assess consistency between use of the different terms and the Rice/Link and Browman terminology. Table 1 lists the term used in each publication and, given the context in which each term was used in each paper, indicates the synonymous term in the Rice/Link and Browman terminology. Four papers used a new term, ecosystem approach to fisheries management, and in each case usage was synonymous with the term EBFM in the Rice/Link and Browman terminology. Two papers used the term EAM, one in the sense of EBM and one in the sense of EAF in the Rice/Link and Browman terminology.

Table 1. Assessing consistency between the terms used in 100 randomly selected papers published from 2006 onwards and the terminology established from Rice's (2011) and Link's and Browman's (2014) papers (Figure 1). The term used in each publication is indicated in the row header (left column) and, given the context of usage in the publication, the equivalent term in the Rice/Link and Browman terminology is indicated as the column header (2nd row).

Term in publication	Synonymous term in Rice (2011) / Link and Browman (2014) terminology (Figure 1)		
	EAF	EBFM	EBM
EAF	Kifani *et al.* 2008 Jarre *et al.* 2008 Fromentin *et al.* 2009 Gasalla *et al.* 2009 Casini *et al.* 2010[4] Carvalho *et al.*, 2011 Cambie *et al.* 2012 Cardinale and Helm 2012 Gascuel *et al.* 2012 Okes *et al.* 2012 Rijnsdorp *et al.* 2012 Avadi and Freon 2013	Hiddink *et al.* 2006 Pope *et al.* 2006 Graham *et al.* 2007 Hill *et al.* 2007 Piet *et al.* 2007 Plagányi *et al.* 2007 Blanchard *et al.* 2009[3] Reiss *et al.* 2009 Kempf 2010 Paterson *et al.* 2010 Reiss *et al.* 2010a Shepherd *et al.* 2011 Bertrand *et al.* 2012 Houle *et al.* 2012 Duggan *et al.* 2013 Fung *et al.* 2013 Hara 2013 Moore 2013 Jennings *et al.* 2014 Lambert *et al.* 2014 Maravelias *et al.* 2014 Modica *et al.* 2014	Dickey-Collas 2014
EBFM	Field *et al.* 2006 Sanchirico *et al.* 2006 Ainsworth *et al.* 2008 Froese *et al.* 2008 Lindegren *et al.* 2009 Casini *et al.* 2010[4] Garrison *et al.* 2010 Spiers *et al.* 2010 Bellchambers *et al.* 2014 Persson *et al.* 2014	Astles *et al.* 2006 Field and Francis 2006 Fletcher 2006[5] Hall *et al.* 2006 Francis *et al.* 2007 Marasco *et al.* 2007 McShane *et al.* 2007 Smith *et al.* 2007 Gaichas 2008 Gremillet *et al.* 2008 Blanchard *et al.* 2009[3] Appollonio 2010 Fletcher *et al.* 2010 Gaichas *et al.* 2010 Gibbs 2010 Zhou *et al.* 2010 Essington and Punt 2011 Hilborn *et al.* 2011 Kellner *et al.* 2011	

		Kim and Zhang 2011 Olson 2011 Emery *et al.* 2012 Jin *et al.* 2012 Nguyen *et al.* 2012 Smale *et al.* 2012 Buchheister *et al.* 2013 Large *et al.* 2013 Fulton *et al.* 2014 NOAA 2014 Pew Trust 2014 Möllman *et al.* 2014[2] Shelton *et al.* 2014 Van Putten *et al.* 2014 Vos *et al.*, 2014 Brown *et al.* 2015	
EBM		Frederiksen *et al.* 2008 Möllman *et al.* 2014[2]	Fletcher 2006[5] Leslie and McLeod 2007 Crowder and Norse 2008 Douvere 2008 Levin and Lubchenco 2008 Ruckleshaus *et al.* 2008 Levin *et al.* 2009 Curtin and Prellezo 2010 Halpern *et al.* 2010 Tallis *et al.* 2010 Fanning *et al.* 2011 Agardy *et al.* 2011 Espinosa-Romero *et al.* 2011 Samhouri and Levin 2012 Levin *et al.* 2014 Samhouri *et al.* 2014 Walther and Möllman 2014[1]
EAM	Hollowed *et al.* 2011		Walther and Möllman 2014[1]
EAFM[6]		Bellido *et al.*, 2011 Gascuel *et al.* 2012 Kraak *et al.* 2012 Babouri *et al.* 2014	

1. Walther and Möllman (2014) appear to use the terms EAM and EBM interchangeably.

2. Möllman *et al.* (2014) appear to use the terms EBFM and EBM interchangeably.

3. Blanchard *et al.* (2009) appear to use the terms EBFM and EAF interchangeably.

4. Casini *et al.* (2010) use the term EBFM in their key-words, but use the term EAF throughout the text.

5. Fletcher (2006) uses both EBFM and EBM and uses both in the same sense as Link and Browman (2014) (Figure 1).

6. Some papers used the term ecosystem approach to fisheries management (EAFM)

Figure 3 illustrates the degree of consistency in use of the terms EAF, EBFM and EBM between the Rice/Link and Browman terminology and the 100 papers. The higher the development level of the EAM, the greater the consistency in term usage. Thus, in 90% of papers that used the term EBM, usage was consistent with the Rice/Link and Browman terminology. Where usage was inconsistent, the paper used the term EBM in the sense that Link and Browman (2014) apply to the term EBFM, i.e. one EAM development level lower. Among papers that used the term EBFM, 78% used it in the same sense as Link and Browman (2014); the remaining papers all used EBFM in the same sense that Link and Browman (2014) apply to the term EAF, again one EAM development level lower. Use of the term EAF was much less consistent; only 34% of papers used this term in the same sense that Link and Browman (2014) do. In this instance, the majority of papers (63%) used the term EAF in the sense that Link and Browman (2014) apply to the term EBFM: one level higher in the EAM development process.

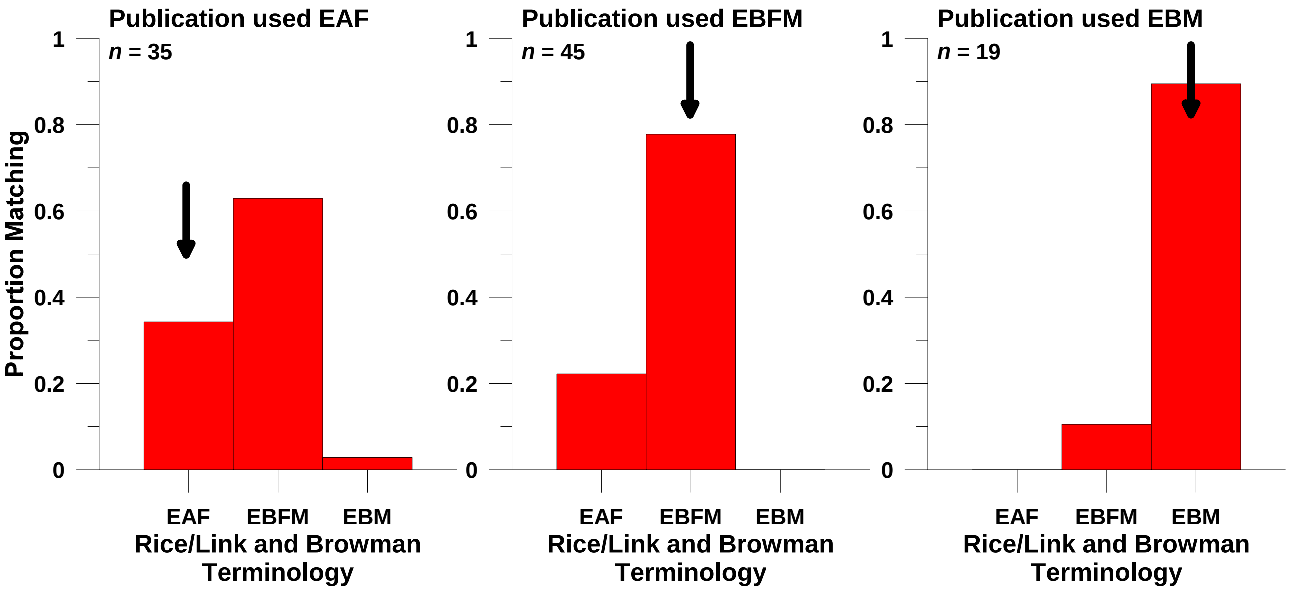


Figure 3. Consistency in the use of the terms EAF, EBFM and EBM in 100 randomly selected papers published since 2006 and the terminology established following Rice (2011) and Link and Browman (2014) (Figure 1). Arrows indicate the term used in the publication. The sample size, the number of publications using the term, in each case is indicated (n).

Our scrutiny of 100 recent EAM publications revealed a notable inconsistency. Of the 105 usages of the different terms identified, only 19 (18%) related to use of the term EBM; a proportion considerably smaller than anticipated given the increase in the publication rate of papers using this term observed in recent years (Figure 2). In many cases, papers used the term EBM in the title, abstract or keywords, then subsequently used a different term, more often than not EBFM, throughout the text. In other instances, papers with EBM in the title, abstract or keywords subsequently extended the term in the text, thus ecosystem-based management *of fisheries*, and used it in the sense of EBFM. Both these situations greatly inflated the apparent annual publication rate of papers using the term EBM in the overall Web of Science meta-analysis, while in our scrutiny of 100 papers, such papers were treated as if they had used the term EBFM. Some papers also use the terms EBM, EBFM, and ecosystem-based management *of fisheries* interchangeably and always synonymously with use of the term EBFM by Link and Browman (2014). This further highlights the need for the establishment of a consistent EAM terminology, which then needs to be used consistently, particularly by those scientists working at the interface of science and policy. One paper reviewed used four different terms, all apparently meant in the sense attributed to EBM by Link and Browman (2014).

Several of these recent papers provided support to the construct of the EAM implicit in the Rice/Link and Browman terminology illustrated in Figure 1. Gaichas (2008) considers EBFM to represent an extension of the concept of sustainable fisheries, not just for target stocks, but for the whole ecosystem. Marasco *et al.* (2007) suggests that the transition to EBFM needs to be an evolutionary process, rather than a revolutionary one. Fletcher (2006) explicitly states that the EAM consists of a hierarchy of management levels and, in a construct remarkably similar to that of Link and Browman (2014), identifies the EBFM level as applying to just the single fisheries sector while EBM applies similar management ideals to multiple sectors. Nguyen (2012) also considers that EBFM and EBM relate to very different, but complementary, governance systems; the former considering just the fisheries sector, while the latter address multiple sectors. And, congruent with these ideas, Möllman *et al.* (2014) consider EBFM to be a component of EBM.

Conclusions

Our review of the EAM literature may not be exhaustive, but we believe that it is a representative sample of the majority of papers published on the subject. Arkema *et al.* (2006) concluded that all the different terms in the literature relating to the EAM of marine natural resources were used with essentially similar meanings in mind. Our review does not lead to the same conclusion. More recently, as scientific knowledge has expanded over time, the underlying concept of the EAM has evolved, and our understanding of exactly what each phase in this evolutionary process entails has increased. This has crystallised into a full description of the process, with the application of specific terms to each stage of the process, provided by Link and Browman (2014). Our review of the literature suggests that whatever term is used in the specific paper under review, it can be related to one or other of the four main phases in the evolution of the EAM, and is therefore synonymous with the particular term applied to that specific phase by Link and Browman (2014). For the remainder of this report therefore, we use the terms EAM, CFM, EAF, EBFM and EBM in the same sense that Rice (2011) and Link and Browman (2014) apply to these terms, and in the sense illustrated in Figure 1. Furthermore we suggest that this terminology could be adopted for the future, to promote clearer communication within the scientific community, between scientists and policy makers, and between the management community and the various stakeholders who use marine natural resources.

Aims and objectives of the MSFD

Our review of the EAM literature above was prompted by the need to determine what the MSFD meant by "ecosystem-based approach to the management of marine natural resources". Now that we have adopted a specific terminology, we return to this question. Our purpose here is to thoroughly review the MSFD documentation to understand exactly what its goals, aims and objectives are, what methods it proposes to achieve these, and in doing so to determine exactly what sort of EAM is required within the context of the Rice/Link and Browman terminology established in Figure 1.

Articles 8, 43 and 44 of the preamble makes it clear that implementation of the MSFD requires an EAM that will manage human activities in such a way as to ensure sustainable use of marine goods and services, while at the same time maintaining good environmental status (GES) through protection and preservation of the marine environment so as to prevent decline in environmental status. More than this, where the marine environment has been adversely affected by past human activity, MSs should take action to restore the health of the marine environment (EC, 2008). Chapter 1, the "General Provisions" of the MSFD, formalises some of this earlier preamble; Article 1 explicitly sets out the directive's goals (EC, 2008). These can be summarised as requiring MSs to develop strategies to protect and preserve the marine environment, prevent further deterioration, and where feasible restore adversely affected marine ecosystems. These strategies should involve an ecosystem-based approach to the management of human activities to ensure that the collective pressure of such activities is compatible with achieving GES, that the capacity of marine ecosystems to respond to human-induced changes is not compromised, while still enabling the sustainable use of marine goods and services by present and future generations.

Paragraph 25 of the MSFD preamble states "*… …, Member States should then determine for their marine waters a set of characteristics for good environmental status … …*" (EC, 2008). Article 3 provides definitions of some of the terms used, thus paragraph 4 defines 'environmental status' as "*the overall state of the environment in marine waters, taking into account the structure, function and processes of the constituent marine ecosystems together with natural physiographic, geographic, biological, geological and climatic factors, as well as physical, acoustic and chemical conditions, … …*" and paragraph 5 expands on this by defining 'good environmental status' as "*the environmental status of marine waters where these provide ecologically diverse and dynamic oceans and seas which are clean, healthy and productive within their intrinsic conditions, and the use of the marine environment is at a level that is sustainable, thus safeguarding the potential for uses and activities by current and future generations, i.e.: (a) the structure, functions and processes of the constituent marine ecosystems, together with the associated physiographic, geographic, geological and climatic factors, allow those ecosystems to function fully and to maintain their resilience to human-induced environmental change. Marine species and habitats are protected, human-induced decline of biodiversity is prevented and diverse biological components function in balance; (b) hydro-morphological, physical and chemical properties of the ecosystems, including those properties which result from human activities in the area concerned, support the ecosystems as described above. Anthropogenic inputs of substances and energy, including noise, into the marine environment do not cause pollution effects*" (EC, 2008). This could not be clearer. The status of all components of marine ecosystem must be satisfactory if overall GES is to be achieved, not just the status of species targeted by fisheries, or any other specific components subject to directed human exploitation or activity.

Paragraph 5 of article 3 concludes "*Good environmental status shall be determined … … on the basis of the qualitative descriptors in Annex I*". These eleven qualitative descriptors listed in Annex I are shown in Table 2, and again the message could not be clearer; the MSFD requires GES be achieved across all components of the marine ecosystem. Descriptor 3 does require the populations of species targeted by fisheries to be in a satisfactory state, but Descriptor 1 addresses the maintenance of biodiversity, requiring the quality and occurrence of habitats and the distribution and abundance of species to be at good environmental status. Descriptor 4 requires *all* elements of food webs to be in a state consistent with satisfactory food web structure and function. Descriptor 6 requires that the state of the seafloor habitat is adequate to maintain ecosystem structure and functionality, especially the benthic ecosystem.

Table 2. The Qualitative Descriptors of Good Environmental Status listed in Annex 1 of the MSFD (EC, 2008).

No.	Qualitative Descriptor of GES
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.
2	Non-indigenous species introduced by human activities are at levels that do not adversely alter the ecosystems.
3	Populations of all commercially exploited fish and shellfish are within safe biological limits, exhibiting a population age and size distribution that is indicative of a healthy stock.
4	All elements of the marine food webs, to the extent that they are known, occur at normal abundance and diversity and levels capable of ensuring the long-term abundance of the species and the retention of their full reproductive capacity.
5	Human-induced eutrophication is minimised, especially adverse effects thereof, such as losses in biodiversity, ecosystem degradation, harmful algae blooms and oxygen deficiency in bottom waters.
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.
7	Permanent alteration of hydrographical conditions does not adversely affect marine ecosystems.
8	Concentrations of contaminants are at levels not giving rise to pollution effects.
9	Contaminants in fish and other seafood for human consumption do not exceed levels established by Community legislation or other relevant standards.
10	Properties and quantities of marine litter do not cause harm to the coastal and marine environment.
11	Introduction of energy, including underwater noise, is at levels that do not adversely affect the marine environment.

Paragraph 24 of the MSFD preamble states "*As a first step in the preparation of programmes of measures, Member States across a marine region or subregion should undertake an analysis of the features or characteristics of, and pressures and impacts on, their marine waters, identifying the predominant pressures and impacts on those waters, … …*". Paragraph 1 of article 8 provides the detail, "*States shall make an initial assessment of their marine waters, taking account of existing data where available and comprising the following: (a) an analysis of the essential fea-*

tures and characteristics, and current environmental status of those waters, based on the indicative lists of elements set out in Table 1 of Annex III, and covering the physical and chemical features, the habitat types, the biological features and the hydro-morphology; … …" This list of elements of physical, biological and chemical features given in Table 1 of Annex III is again repeated here (Table 3). This list is extensive and explicit in listing the features, characteristics and components of marine ecosystems to which the MSFD applies and for which MSs are obliged to achieve GES. Just about all components of marine ecosystems imaginable are listed here, again reinforcing the point that the MSFD applies to the whole marine ecosystem; all parts of the marine ecosystem need to be in a satisfactory state to meet fully the aspirations of the MSFD.

Table 3. The indicative list of characteristics of marine ecosystems to which the MSFD applies provided in Table 1 of Annex III of the MSFD document (EC, 2008).

Main Category	Examples of Characteristics
Physical and chemical features	Topography and bathymetry of the seabed
	Annual and seasonal temperature regime and ice cover, current velocity, upwelling, wave exposure, mixing characteristics, turbidity, residence time
	Spatial and temporal distribution of salinity
	Spatial and temporal distribution of nutrients (DIN, TN, DIP, TP, TOC) and oxygen
	pH, pCO2 profiles or equivalent information used to measure marine acidification
Habitat types	The predominant seabed and water column habitat type(s) with a description of the characteristic physical and chemical features, such as depth, water temperature regime, currents and other water movements, salinity, structure and substrata composition of the seabed
	Identification and mapping of special habitat types, especially those recognised or identified under Community legislation (the Habitats Directive and the Birds Directive) or international conventions as being of special scientific or biodiversity interest
	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
Biological features	A description of the biological communities associated with the predominant seabed and water column habitats. This would include information on the phytoplankton and zooplankton communities, including the species and seasonal and geographical variability
	Information on angiosperms, macro-algae and invertebrate bottom fauna, including species composition, biomass and annual/seasonal variability
	Information on the structure of fish populations, including the abundance, distribution and age/size structure of the populations
	A description of the population dynamics, natural and actual range and status of species of marine mammals and reptiles occurring in the marine region or subregion
	A description of the population dynamics, natural and actual range and status of species of seabirds occurring in the marine region or subregion
	A description of the population dynamics, natural and actual range and status of other species occurring in the marine region or subregion which are the

	subject of Community legislation or international agreements
	An inventory of the temporal occurrence, abundance and spatial distribution of nonindigenous, exotic species or, where relevant, genetically distinct forms of native species, which are present in the marine region or subregion
Other features	A description of the situation with regard to chemicals, including chemicals giving rise to concern, sediment contamination, hotspots, health issues and contamination of biota (especially biota meant for human consumption)
	A description of any other features or characteristics typical of or specific to the marine region or subregion

Paragraph 24 of the MSFD preamble quoted above also refers to the *"pressures and impacts"* on marine ecosystems. Fishing is clearly one of the human activities requiring management in order to meet MSs' obligations under the MSFD. Annex 1 includes a Descriptor of GES, Descriptor 3, which specifically requires the targeted stocks to meet a specified condition if GES is to be achieved (Table 2), but this is made even more explicit elsewhere in the MSFD. For example, article 39 of the preamble states "*Measures regulating fisheries management can be taken in the context of the Common Fisheries Policy, … …, based on scientific advice with a view to supporting the achievement of the objectives addressed by this Directive, including the full closure to fisheries of certain areas, to enable the integrity, structure and functioning of ecosystems to be maintained or restored … …*" and article 40 of the preamble states "*The Common Fisheries Policy, including in the future reform, should take into account the environmental impacts of fishing and the objectives of this Directive*". The MSFD requires fisheries to be managed in such a way as to achieve healthy states for each of the targeted stocks. Within the terminology established in the previous section, the ecosystem-based approach required under the MSFD is therefore at the least an EAF. However, the MSFD clearly requires more than this, it requires that fisheries be managed in such a way as to not compromise the GES of all other components of the marine ecosystem. This raises the management approach level required to at least that of EBFM.

But fisheries are not the only human activity of concern under the MSFD. Descriptor 8 for example clearly requires pollution to be strictly controlled so as to reduce contaminants in the marine environment to negligible or low levels of (Table 2). Descriptor 9 emphasises the need to further ensure that pollution related contaminant levels are sufficiently low as to not build up within parts of the food chain that ultimately lead to human consumption (Table 2). The need to control or eliminate pollution is also explicitly stipulated at several other places in the MSFD; for example in paragraph 2 of Article 1 "*… prevent and reduce inputs in the marine environment, with a view to phasing out pollution …*", again in paragraph 5 of article 3 "*… Anthropogenic inputs of substances and energy, including noise, into the marine environment do not cause pollution effects … …*", and once more in paragraph 8 of article 3 "*'pollution' means the direct or indirect introduction into the marine environment, as a result of human activity, of substances or energy, including human-induced marine underwater noise, which results or is likely to result in deleterious effects … …*". The two excerpts from article 3 cited above also explicitly require that the input of under-water noise into the marine environment linked to human activity also needs to be managed so as to minimise its impact on marine ecosystems. Several human activities, in addition to fishing, introduce noise into the marine environment: notably seismic survey, pile-driving associated with offshore wind-farms, and gravel extraction. That these activities will have to be managed so that the deleterious effects of introduced noise are kept to a minimum is further emphasised by the inclusion of a Descriptor of GES, Descriptor 11, explicitly to that effect (Table 2). Shipping is also a major potential vector for the introduction of non-indigenous species (Carlton 1985; Williams *et al.* 1988; Hallegraff, 1998; Shiganova and Bulgakova, 2000; Gollasch and Rosenthal, 2006; Werschkun *et al.*, 2014), and again another Descriptor of GES, Descriptor 2, explicitly requires that this activity also be managed appropriately (Table 2).

Paragraph 1 of article 8, already part quoted above, goes on " … …; *(b) an analysis of the predominant pressures and impacts, including human activity, on the environmental status of those waters which: (i) is based on the indicative lists of elements set out in Table 2 of Annex III,* … …". This list of elements of human activity, pressures and impacts given in Table 2 of Annex III is again repeated here (Table 4), and from this extensive list, it is clear that the MSFD is intended to ensure that all human activities in the marine environment involving exploitation of marine natural resources (with a few explicit exceptions such as military use) are regulated and restricted to levels consistent with sustainable use. Paragraph 1 of article 9 underlines the obligation for MSs to ensure that human pressures and impacts in the marine environment should not compromise GES and paragraph 1 of article 10 requires MSs to ensure that environmental targets that represent GES take account of the indicative list of pressures and impacts (Table 4). The MSFD is clearly not just a tool to ensure that fisheries are regulated so as to reduce unwanted impacts on the broader marine ecosystem; it is clearly intended to ensure that all marine activities are managed in such a way as to promote sustainable use of marine resources, but at the same time safeguarding the capacity of marine ecosystems to deliver all the goods and services that society might require into the future. The MSFD applies to the whole marine ecosystem and to all sectors of society that operate in the marine environment; meeting its goals and objectives will require full EBM as we have defined this term above.

Table 4. The indicative list of pressures and impacts on marine ecosystems to which the MSFD applies provided in Table 2 of Annex III of the MSFD document (EC, 2008).

Pressure	Examples of Impact
Physical loss	Smothering (e.g. by man-made structures, disposal of dredge spoil)
	Sealing (e.g. by permanent constructions)
Physical damage	Changes in siltation (e.g. by outfalls, increased run-off, dredging/disposal of dredge spoil)
	Abrasion (e.g. impact on the seabed of commercial fishing, boating, anchoring)
	Selective extraction (e.g. exploration and exploitation of living and non-living resources on seabed and subsoil)
Other physical disturbance	Underwater noise (e.g. from shipping, underwater acoustic equipment)
	Marine litter
Interference with hydrological processes	Significant changes in thermal regime (e.g. by outfalls from power stations)
	Significant changes in salinity regime (e.g. by constructions impeding water movements, water abstraction)
Contamination by hazardous substances	Introduction of synthetic compounds (e.g. priority substances under Directive 2000/60/EC which are relevant for the marine environment such as pesticides, antifoulants, pharmaceuticals, resulting, for example, from losses from diffuse sources, pollution by ships, atmospheric deposition and biologically active substances)

	Introduction of non-synthetic substances and compounds (e.g. heavy metals, hydrocarbons, resulting, for example, from pollution by ships and oil, gas and mineral exploration and exploitation, atmospheric deposition, riverine inputs)
	Introduction of radio-nuclides
Systematic and/or intentional release of substances	Introduction of other substances, whether solid, liquid or gas, in marine waters, resulting from their systematic and/or intentional release into the marine environment, as permitted in accordance with other Community legislation and/or international conventions.
Nutrient and organic matter enrichment	Inputs of fertilisers and other nitrogen and phosphorus-rich substances (e.g. from point and diffuse sources, including agriculture, aquaculture, atmospheric deposition)
	Inputs of organic matter (e.g. sewers, mariculture, riverine inputs)
Biological disturbance	Introduction of microbial pathogens
	Introduction of non-indigenous species and translocations
	Selective extraction of species, including incidental non-target catches (e.g. by commercial and recreational fishing)

Integration – the key to EBM

Each incremental step in the development of the EAM involves increasing levels of integration. Moving from CFM to an EAF requires integration of the management objectives for different stocks. Stepping up to EBFM requires fisheries objectives to be weighed against the need to preserve particular aspects of ecosystem structure and function. The final step from EBFM to full EBM requires the aspirations and ecological impacts of all sectors operating in the marine environment to be balanced, one against the other, in order to achieve the optimal compromise that best meets the needs of all sectors, whilst still protecting ecosystem integrity. Misund and Skjoldal (2005) assert that since nature is an integrated entity, a holistic integrated EAM is required to manage nature. Rice (2011) summed this up in listing integrated management as the fourth component in the development of an EAM. Rice makes it clear that integrated management and EBM are not the same thing, rather integrated management is a key aspect of successful EBM.

However, as with the original development of the EAM and its associated diverse and confusing terminology, it is not exactly clear what needs to be integrated and how this integration should be achieved?

- Firstly, some authors consider that the science process underpinning the provision of advice to managers requires integration; a multidisciplinary scientific approach is necessary (Cury 2004) in order to facilitate greater **integration of scientific knowledge** across different ecosystem components and provide the integrated advice required to support EBM (Caddy and Seijo, 2005Livingston, 2005). The creation of integrated advisory bodies may facilitate this, since the whole science process, not just the final science products, needs better integration (Rice 2005), and PICES and ICES both seem to be progressing along such lines (Livingston, 2005; Misund and Skjoldal, 2005). This should also promote closer integration between the scientific and management processes that other authors consider essential for successful EBM (Cury, 2004; Sherman *et al.*, 2005).

- Secondly, many authors consider the **integration of policy objectives**, such as fisheries objectives and conservation needs, to be one of the key elements of EBM. This has to build on the integration of scientific knowledge, for example knowledge of target species population dynamics as well as knowledge of non-target species and ecosystem structure to understand the dynamics of exploited systems (Zeller and Pauly, 2004; caddy and Seijo, 2005; Tudela and Short, 2005; Misund and Skjoldal, 2005). Others expand this stance to include other sectors that exploit marine natural resources; the need to balance the goals of all sectors operating within the marine environment against the obligation to protect the marine ecosystem that provides these resources by ensuring that such exploitation is sustainable (Agardy, 2005; Valdimarsson and Metzner, 2005).
- Thirdly, some authors focus less on integrating the objectives of different sectors and instead concentrate on the need to address the interactive effects of multiple pressures on the ecosystem associated with different sector's activities (Lotze, 2004; Rosenberg and McLeod, 2005; Misund and Skjoldal, 2005): akin to **integration of multiple impacts**.
- Finally, other authors point out the need for improved **integration of the planning and management processes** that regulate the various human activities so as to ensure that all sectors operating in the marine environment are managed appropriately; that their combined impacts are kept within sustainable bounds (Rice, 2005). Such integration should also help to create more cohesive management policies that are more comprehensible to the different sectors involved (Rosenberg and McLeod, 2005).

Successful implementation of EBM clearly requires integration at many levels: integration of knowledge from biology, oceanography, economics and other social sciences, law and politics (Sissenwine and Murawski, 2004) and it will include stakeholders, social and political scientists, economists, lawyers, political lobbyists, educators, journalists, civil engineers, ecologists, fishery scientists and oceanographers, all operating in a conciliatory and integrative environment (Browman and Stergiou, 2004a). One of the primary benefits of fully integrated EBM is that it should offer more complete and balanced accounting of the full range benefits and costs to society associated with developing sustainable approaches for living marine resources (Sissenwine and Murawski, 2004). However, some authors still sound a note of caution. Jennings (2004) believes that "The most effective progress towards an ecosystem approach is likely to be achieved by moving forward collectively, integrating the useful aspects of existing approaches into new ones". Mace (2004) shares this sentiment adding "We need to develop ecosystem-based approaches to fisheries that build upon and integrate 'traditional' single-species objectives, not solutions that abandon traditional approaches that have never been fully implemented, in favour of what are often ill-defined concepts that may do little to solve the overall problems and may not be operational". Rice (2005) also suggests that we should use what we know, not rush to build and use a brand new tool kit.

Despite these cautionary reminders, recent definitions of EBM (or the EAM) all explicitly incorporate the concept of integration. ICES (2000) proposed the following definition for an EAM: the "*integrated management of human activities based on knowledge of ecosystem dynamics to achieve sustainable use of ecosystem goods and services, and maintenance of ecosystem integrity*". This formed the basis for the technical definition used in the statement from the First Joint Ministerial Meeting of the Helsinki (HELCOM) and Oslo/Paris Commissions (OSPAR) in Bremen 2003: "*The comprehensive integrated management of human activities based on the best available scientific knowledge about the ecosystem and its dynamics, in order to identify and take action on influences which are critical to the health of marine ecosystems, thereby achieving sustainable use of ecosystem good and services and maintenance of ecosystem integrity*" (Misund and Skjoldal, 2005). Similarly, over 200 scientists from institutions across the USA agreed a third definition of EBM: "*Ecosystem-based management is an integrated approach to management that considers the entire ecosystem, including humans. The goal of ecosystem-based management is to maintain an ecosystem in a healthy, productive and resilient condition so that it can provide the services humans want and need. Ecosystem-based management differs from current approaches that usually focus on a single species, sector, activity or concern; it considers the cumulative impacts of different sectors*" (Rosenberg and McLeod, 2005).

All three definitions stress the need for integrated management of human activities, implying that integration between different sectors of society using marine natural resources is the key element of EBM (Misund and Skjoldal, 2005). The Norwegian government's white paper on a 'Clean and Rich Sea' sees EBM as the means of achieving better sector integration (Misund and Skjoldal, 2005). To achieve this, Misund and Skjoldal (2005) suggest that assessment of the combined impacts on marine ecosystems from different sectors will be necessary, and in order to maximise the effectiveness of EBM, the different sectors will have to work closely together; collaboration between the fisheries and conservation sectors, for example, will be essential. Tudela and Short (2005) view this from an opposite perspective, suggesting that adopting EBFM provides an opportunity for achieving high levels of integration between fisheries objectives and conservation goals. Rosenberg and McLeod (2005) also emphasise the need for EBM to account for interactions among sectors by integrating management across multiple sectors. They point out that neither the impacts from, nor the management policies for different sectors are independent of one another.

Integration is required by the MSFD

In applying EBM to implement the MSFD, the need for integration is clear (EC, 2008). Article 21 of the preamble, which states "*It is crucial for the achievement of the objectives of this Directive to ensure the integration of conservation objectives, management measures and monitoring and assessment activities … …*", clearly requires integration of the various conservation goals for different components of the marine ecosystem. Paragraph 3 of Article 1 implies the need for integration in that it requires MSs to ensure "*that the collective pressure of such* (human) *activities is kept within levels compatible with the achievement of good environmental status …*", suggesting that it is the cumulative impacts of all human activities that must be managed and kept collectively to sustainable levels. Similarly, paragraph 1 of article 8, quoted in part twice above, continues "*… … and covers the qualitative and quantitative mix of the various pressures, as well as discernible trends; … … ii) covers the main cumulative and synergetic effects; … …*" also implicitly infers the need for integration since it requires that the cumulative and synergistic effects of multiple pressures be properly accounted for. However paragraph 4 of Article 1 is explicit, stating "*This Directive shall contribute to coherence between, and aim to ensure the integration of environmental concerns into, the different policies, agreements and legislative measures which have an impact on the marine* *environment*". Articles 39 and 40 of the preamble explicitly identify the Common Fisheries Policy as one of the policies considered under this statement.

Development of the assessment process to meet EAM needs

The need to monitor change in different attributes of a range of ecosystem components is a core requirement of EBM (Link, 2005; Rice, 2005). This has stimulated the development of numerous 'ecological indicators' (Link, 2005; Daan *et al.*, 2005; Shannon *et al.*, 2010; Shin and Shannon, 2010; Shin *et al.*, 2010a; Borja *et al.*, 2011; Greenstreet *et al.*, 2012b). This proliferation in the number of indicators potentially available quickly led to the introduction of criteria to assess the performance of indicators and inform the selection of indicators that would be most useful within a management context (ICES, 2000; Rice and Rochet, 2005; Rochet and Rice, 2005; Piet *et al.*, 2008). Indicators can only directly underpin management if the relationship between pressure and state is clearly understood; only then can the correct management measures be imposed to alter pressure in a particular was as to bring about the desired change in state (Jennings, 2005; Piet and Jennings, 2005). This requirement for there to be a strong and well defined relationship between variation in any proposed 'ecological indicator' and changes in pressure permeates all the sets of criteria used to inform indicator selection (Greenstreet, 2008). This led, for example, to selection of the large fish indicator (LFI) to support the OSPAR Ecological Quality Objective for the North Sea demersal fish community, because this indicator had been especially formulated to render it particularly sensitive to the detrimental impacts of fishing on the community (Greenstreet *et al.*, 2011). Variation in this indicator, and its relationship with fishing mortality, has subsequently been explored in a number of other marine regions (Shepherd *et al.*, 2011b; Modica *et al.*, 2014).

Assuming a given 'ecological indicator' meets the criteria for a good indicator, it is a relatively straightforward task to monitor variation in the indicator and, based on the known relationship to pressure linked to a particular human activity, to use observed changes in the indicator as the basis for advice as to how the human activity should be managed to meet specified targets for the indicator. This process essentially describes the assessment procedure for a single ecosystem component affected by a pressure from a single human activity and has been the basis for the single stock assessments that have underpinned CFM for decades. Target species spawning stock biomass (SSB; the state indicator) is monitored as fishing mortality (the pressure indicator) varies. Fishing mortality rates can be managed, either by placing limits on the total catch allowed to fishermen (catch restriction), or by restricting the amount of time that they are allowed to fish (effort restriction), so as to achieve specified targets for SSB (Rice, 2009). However, it has generally been assumed that different attributes (e.g. species diversity, mean weight, mean age at maturity, etc) of any single ecosystem component might vary independently (Fulton *et al.*, 2005; Piet and Jennings, 2005; Greenstreet and Rogers, 2006), so that to account for change in all attributes of a single ecosystem component requires application of a suite of 'ecological indicators' (Blanchard *et al.*, 2010; Bundy *et al.*, 2010). For example, Greenstreet *et al.* (2012b) suggest that a suite of seven indicators is necessary to ensure that all changes in the general health of the North Sea demersal fish assemblage are adequately covered. Assessment of the state of an ecosystem component monitored using a suite of 'ecological indicators' presents a more complex problem requiring the information obtained from several different indicators to be integrated or aggregated in some way, following a predefined procedure, to derive an overall impression of the state of the ecosystem component in question (Rochet *et al.*, 2005; Shin *et al.*, 2010b; Shannon *et al.*, 2010).

From this it is clear that at each step in the development of the EAM, the there is an increase in the amount of integration necessary in the assessment process on which management advice depends. In single stock assessments to support CFM no integration is required. In the step from CFM to an EAF some integration becomes necessary as part of the management strategy evaluation process by which the optimal fishing strategy to maximise catches across several stocks might be determined (Mackinson *et al.*, 2009; Kraak *et al.*, 2010; Ulrich *et al.*, 2011; Rijnsdorp *et al.*, 2012). The step from an EAF to EBFM then involves a further step-wise increase in the level integration involved in the assessment process. For example, if using the LFI to monitor the detrimental impact of fishing on the broader fish community, then in addition to the integration involved in the fisheries management strategy evaluation, the effect of each fishing strategy on the LFI has also to be assessed so as to ensure that the strategy selected does not compromise targets set for the LFI. If instead a suite of indicators is used to monitor the health of the fish community, then this process will need to be repeated for each 'ecological indicator' in the suite and an integrated assessment performed across the suite of indicators so that the need to ensure that the overall health of the fish community then influences fishing strategy selection. The addition of further 'ecological indicators' used to monitor change in the state of other ecosystem components, such as seabed habitats, benthic invertebrates, seabirds and marine mammals clearly raises the level of integration required considerably. Moving from EBFM to full EBM raises the complexity of the assessment process exponentially; the process described above for EBFM now has to be repeated for each human activity and the pressures these impose on each ecosystem component (e.g ICES 2006).

The EAM has been criticized as being immature and too complicated to be successfully implemented (Murawski 2007, Walther and Möllmann 2014). Rice (2011) makes a good point when he suggests that the increase in scientific knowledge is incremental, while changes in policy are saltatory in nature (Figure 4). Our increasing understanding of the detrimental impacts of human activities, especially fishing, on the wider marine environment and ecosystems in European waters through the latter decades of the 20th century and into the start of the 21st century was finally sufficient in 2008 to bring about a change in management policy and the MSFD was introduced. Up to this point, bringing about this change in management attitude was perhaps the principal motivation underlying much of the marine science carried out. After 2008, however a change in scientific focus became necessary; now scientists need to concentrate on accumulating the scientific knowledge required to implement the EBM to meet the objectives of the MSFD; the gap between policy and science needs to be closed. Walther and Möllmann (2014) list some of the challenges that need to be met if EBM is to be successfully implemented. These include the fact that the EAM is poorly defined; we hope the discussion above goes some way to resolving this issue in respect of the MSFD. Other impediments include a lack of routines or protocols to develop the appropriate advice and poor communication between scientists, advisers, and managers. However, Walther and Möllmann (2014) also suggest that development of Integrated Ecosystem Assessment (IEA) protocols to address the problems discussed above should help to resolve these outstanding issues; development of appropriate IEA protocols is the means by which EBM can be made operational (Levin *et al.*, 2009; Samhouri *et al.*, 2010; Samhouri and Levin 2012; Belgrano and Fowler, 2011). Indeed Möllmann *et al.* (2014) imply that it is perhaps the lack of established IEA procedures that has delayed the progression from CFM to EBM in Europe.

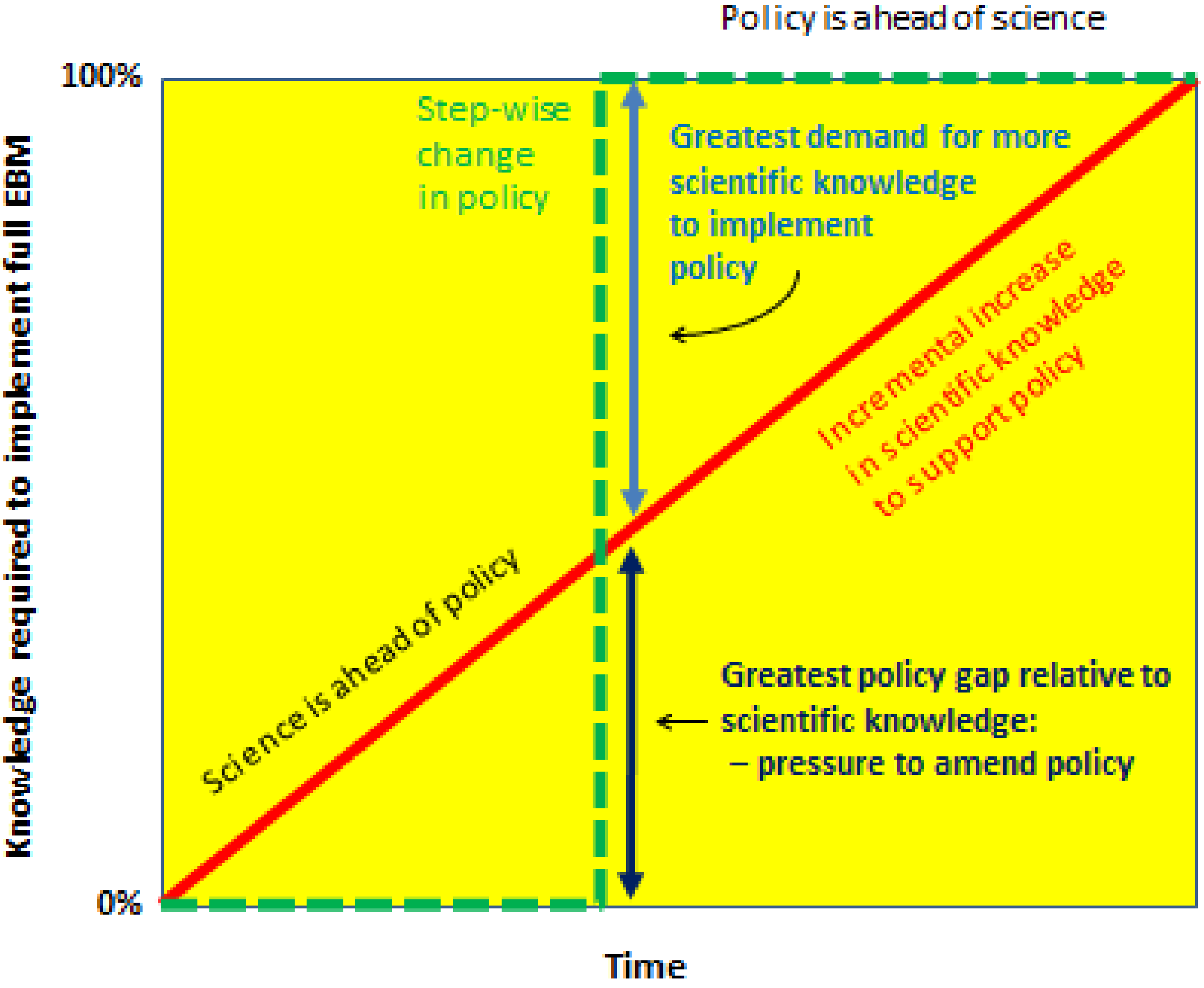


Figure 4. Temporal changes in science knowledge and in policy leading to asynchrony between science support and policy. Policy and science start in complementary states but over time science knowledge accumulates incrementally, whereas policy remains stable. Science gets increasingly ahead of policy as the gap between what is known about, for example, the ecosystem effects of fishing, and what policy requires of fisheries management grows larger. When policy changes to address the knowledge gap, frequently this change is instantaneous and large enough to address the implications of the best-studied cases. This abruptly places policy ahead of science and the gap in knowledge necessary to support implementation of the policy broadly, not just in the best-studied cases, may be large. After the policy change, the gap closes incrementally as knowledge continues to accumulate incrementally (adapted after Rice, 2011).

Integrated ecosystem assessment

Integrated Ecosystem Assessment (IEA) represents a powerful tool for the implementation of EBM (Walther and Möllmann, 2014; Levin *et al.*, 2014; Samhouri *et al.*, 2014). The concept of the IEA has been introduced by Levin *et al.* (2009) as a framework for "*organizing science in order to inform decisions in marine EBM*". Several definitions of IEA have been provided reflecting somewhat personal perspectives of what IEA should be (Dickey-Collas, 2014; Levin *et al.*, 2009). However, all definitions promote the holistic and interdisciplinary nature of IEAs. IEAs synthesize data from all relevant sources to support decision making for achieving predefined management goals (Walther and Möllmann, 2014; Levin *et al.*, 2014). IEAs are holistic and regional, incorporating knowledge on physical, chemical biological and human components within a relevant spatial scale (Levin *et al.*, 2014). Another important aspect of IEA is the embracement of an adaptive management regime. Though not explicitly termed as an IEA, the MSFD required the member states to perform an initial holistic ecosystem assessment by 2012 (Article 8). This initial assessment has to be updated every six years (Article 17). By demanding the assessment of ecological features, human pressures and socioeconomic impacts the MSFD is setting the stage for performing IEA within European policy frameworks (Claussen *et al.*, 2011).

Levin *et al.* (2009) provide a framework for IEA consisting of cyclical six step protocol consisting of: scoping; development of ecosystem indicators and targets; risk analysis; assessment of ecosystem status to EBM goals; management strategy evaluation; and monitoring of ecosystem indicators and management effectiveness.

Scoping

Scoping represents the first step of an IEA cycle. This phase should identify the overarching goals that EBM is supposed to achieve. Relevant spatial scales are determined and focal ecosystem components and key threats should be identified (Levin *et al.*, 2014). The scoping process is likely to be complex as it will involve scientists, policy-makers, managers, and stake-holders, all with different perspectives and holding different views as to what EBM should deliver, and therefore how IEA should be carried out, particularly the type of integration rules that should applied at different stages of the integration process (Dickey-Collas, 2014). Establishing the boundaries of the IEA exercise is a key aspect of the scoping process: identifying the factors and objects, temporal and spatial scales, short- and long-term goals, etc, that the assessment is expected to cover (De Young *et al.*, 2008).

Levin *et al.* (2014) expand on what the scoping stage of an IEA intended to make EBM operational needs to include. An overarching goal, or vision, for the ecosystem should be specified. This then needs to be broken down into separate conceptual and operational objectives (Sainsbury and Sumaila, 2003; O'Boyle and Jamieson, 2006). The spatial scale needs to be defined, which will largely depend on a combination of both biological and political considerations (Levin *et al.*, 2014). Management measures applied need to be related to geographically specified areas, and ideally such areas should be consistent with the boundaries of identified LMEs (Sissenwine and Murawski, 2004; Misund and Skjoldal, 2005). Where these LME are shared by two or more countries, the policies that constitute EBM will have to be trans-boundary in nature. For example, the Benguela Current LME is bordered by Angola, Namibia and South Africa. Getting three countries with such diverse societal objectives to agree to collaborative co-ordinated EBM presents considerable challenges and is a further example of the sort of integration that will be necessary if EBM is to be successfully implemented at spatial scales specified at the scale of LMEs (Sumaila, 2005). Temporal scales will also be necessary – when should management objectives be achieved?

Focal ecosystem components need to be specified. These are the major elements of an ecosystem that can be used to organise relevant information in a limited number of discrete, but not necessarily independent categories. These are often linked to ecological structure and function, fisheries and other human pressures on the ecosystem, protected species, habitat, etc (Levin and Schwing, 2011). Key threats to these ecosystem components need to be identified. Some consideration should also be given to the conceptional basis for the integration process, i.e. how information from different sources (e.g. different indicators) should be aggregated to determine an overall assessment of ecosystem status. Since the eventual IEA outcome is so dependent on which integration rules are applied (Ojaveer and Eero, 2011), it is essential that decisions regarding which rules should be used are made *a priori* of the assessment and not during it when there might be a risk of selecting particular rules in order to gain the desired outcome.

Development of ecosystem indicators and targets

As in all previous assessment programmes, indicators are required to monitor change in the state of ecosystem components. The current state of any ecosystem component needs to be gauged in relation to the condition considered acceptable (Jennings, 2005; Samhouri *et al.*, 2009); this desired state therefore represents the target for any given indicator (Trenkel and Rochet, 2003; Fulton *et al.*, 2005; Methratta and Link, 2006). This is the essence of any management process: it has been the key element in fisheries management (Rice 2009) and in the development of the OSPAR Ecological Quality Objectives (EcoQO) pilot project (Heslenfeld and Enserink, 2008). By achieving all targets for the nominated indicators, the overarching goals of EBM should be achieved. For some data-rich components of marine ecosystems, considerable effort has been expended developing a plethora of different metrics that could potentially fulfil an indicator role (Link, 2005; Daan *et al.*, 2005; Shannon *et al.*, 2010; Shin and Shannon, 2010; Shin *et al.*, 2010a; Borja *et al.*, 2011; Greenstreet *et al.*, 2012b). The suite of indicators selected to support EBM has not only to provide the essential scientific information, thereby maximising their value for policy-makers and managers (Levin *et al.*, 2010), but the information conveyed by these indicators must also be communicable to stake-holders and to society at large (Jennings 2005). To this end several authors have proposed sets of criteria by which to evaluate overall indicator performance and these can be used to inform the selection process to ensure that indicators that best meet these criteria are chosen (ICES, 2000; Rice and Rochet, 2005; Rochet and Rice, 2005; Piet *et al.*, 2008).

Risk analysis

Here the term 'risk analysis' is applied in the sense *"to determine the likelihood that an ecosystem indicator will reach or remain in an undesirable state"* (Levin *et al.*, 2009). A key element of the risk analysis phase of the IEA framework is to identify ecosystem components that are particularly vulnerable to specific pressures, and which are actually subjected to those pressures. There is concern that in attempting to implement EBM as fully as possible, effort might be directed at developing indicators and targets for as near complete a range of ecosystem components and pressures as possible. The resulting number of indicators and targets could then become so large as to render any IEA so complicated that it becomes either impossible, or so difficult to interpret, that advice to managers becomes unreliable (Jennings and Le Quesne 2012).

Most sets of criteria used to assess indicator performance place considerable emphasis on the need for an indicator to be sensitive to particular pressures, and for the relationship between variation in these pressures and the response of the indicators to be well understood (ICES, 2000; Rice and Rochet, 2005; Rochet and Rice, 2005; Piet *et al.*, 2008). Use of such criteria should preclude the selection of indicators that are insensitive to variation in pressure, thus eliminating situations A and B in Figure 5. However, indicators could still be selected that, whilst sensitive to a particular pressure, might in reality only detect change in ecosystem components that are not subjected to the pressure (situation C in Figure 5). Figure 5 shows an adaptation of the risk-analysis graphic adopted by Samhouri and Levin (2012), which illustrates variation in risk within two dimensional sensitivity-exposure space; ‘sensitivity’ is a measure of the susceptibility of an ecosystem component, or an attribute thereof, to variation in a pressure, while ‘exposure’ is a measure of the extent to which the ecosystem component is subjected to the pressure. Other studies have used similar graphics, but with subtly different axis labels. Thus Hobday *et al.* (2011) label their y axis ‘susceptibility’ while the x axis is labeled ‘productivity’ where ‘susceptibility’ is directly equivalent to ‘sensitivity’ but where ‘productivity’ is used as a proxy for the capacity of an ecosystem component, or an attribute thereof, to recover from pressure. Where a risk analysis suggests a situation represented by location C in Figure 5, then the influence of such indicators in the IEA aggregation process can be down-weighted, or the indicator even excluded from the assessment exercise. A risk analysis should primarily be undertaken to ensure that indictors able to detect change in ecosystem components most at risk to detrimental impact from human pressure are definitely included in the IEA (situation D in Figure 5). Application of appropriate indicator performance criteria should already have ensured that these indicators are sensitive to the pressure. These indicators could perhaps be accorded a higher weighting in determining the overall IEA outcome.

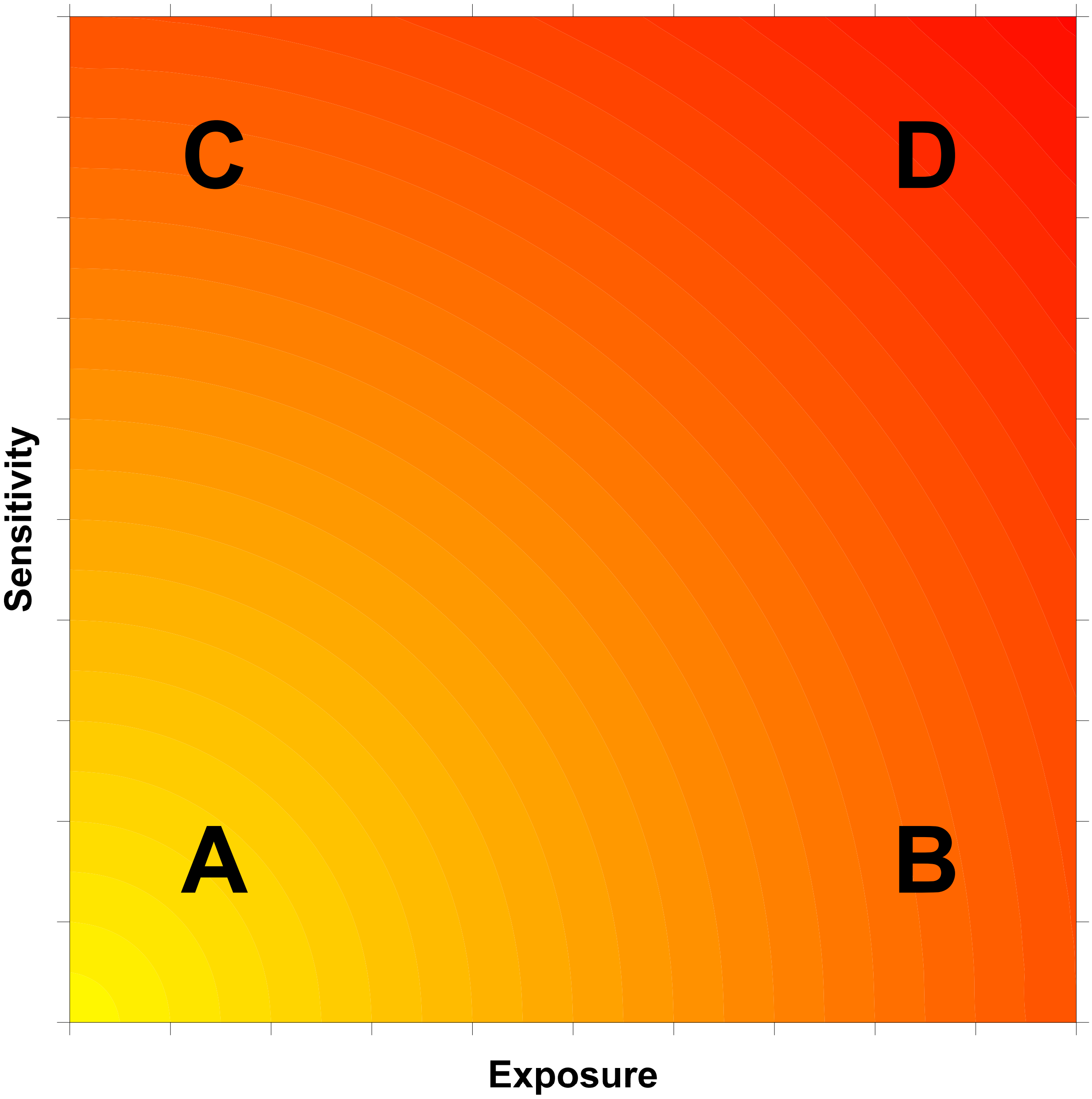


Figure 5. Risk determined as the Euclidean distance of an indicator in exposure-sensitivity space. (A) Depicts the situation of a non-sensitive indicator where exposure to a pressure is minimal. Despite the use of an ineffective indicator, risk is low, but only because exposure to the pressure is negligible. (B) Depicts use of a non-sensitive indicator in a situation where exposure to a pressure exists. In signal-theory terms, such a situation is likely to result in a "miss" (condition assessed as satisfactory when in reality state is below standards set); risk is unknown and could be substantial. (C) Depicts the use of a sensitive indicator but in situation where exposure to pressure is minimal. In signal-theory terms, such a situation is could result in "false-positives" (condition assessed as unsatisfactory when in reality state meets the target); risk is unlikely to be high regardless of the indicator assessment outcome since pressure is negligible. (D) Depicts the use of a sensitive indicator but in situation where exposure to pressure is high. In signal-theory terms, such a situation would generally give "true-positives" (if condition is assessed as unsatisfactory, then this would tend to reflect the true situation); this is the high risk situation (Adapted from Samhouri and Levin, 2012).

Assessment of ecosystem status relative to EBM goals

Having used both a combination of performance criteria to select the most effective indicators and a risk analysis to identify ecosystem components (and hence indicators) most at risk from human pressures to decide the definitive suite of indicators on which the IEA will be based, the next step is to distil out the information conveyed by the suite of indicators to determine whether the high level goals aspired to in applying EBM have been achieved. In most assessments, the first stage in this process involves comparing current indicator values against target values for each indicator independently to address the simple question: 'has the target for the focal indicator been achieved?' This has been the essence of single-species stock assessments that have supported CFM for decades: e.g. 'is fishing mortality less than, and stock biomass greater than, their respective precautionary limits (F_{pa} and B_{pa}) (Rice, 2009)?' A similar approach has been adopted for individual 'ecological indicators'. For example, the LFI is derived using data accrued from groundfish survey monitoring programmes. LFIs have been defined for different marine demersal fish assemblages and compared against area-specific targets stipulated for each assemblage. In each instance the conclusion appears to be that whilst LFI targets have yet to be met, it would appear that the appropriate corrective management measures have been introduced and indicator values are converging on their target value (Greenstreet *et al.*, 2011; Shepherd *et al.*, 2011b; Modica *et al.*, 2014). Similarly, kittiwake (*Rissa tridactyla*) breeding success has been proposed as an indicator of adequate forage fish availability to top predators. Kittiwake chick production is monitored each year at key breeding colonies and can be assessed against either a fixed target of 0.6 chick per breeding pair (Heslenfeld and Enserink, 2008) or a 'moving target' that takes account of inter-annual variation in environmental conditions (Frederiksen *et al.*, 2004).

Where absolute target values for an indicator cannot be specified, but evidence that current status is unacceptable is incontrovertible, then a case can be made for using trends-based targets (Jennings and Dulvy, 2005; Shin *et al.*, 2005). Whilst meeting such targets does not infer that high level goals for EBM in respect of the indicator in question have been achieved, observing a trend in indicator values in the specified direction might at least imply that the restoration process has commenced through the implementation of appropriate corrective management measures (Modica *et al.*, 2014). Trends-based targets have been proposed for 27 sensitive demersal fish species in the North Sea and a probabilistic approach based on the binomial distribution used to determine the number of these species that must meet their trends-based target in order to conclude that appropriate remedial management to restore the populations of vulnerable (to fishing mortality) fish species had been put in place (Greenstreet *et al.*, 2012).

However, EBM requires more than simply comparing numerous separate indicators against their various individual targets; it requires the integration of the information conveyed by all these individual indicator assessments in order to draw overall holistic conclusions regarding the state of the whole ecosystem. How this integration is carried out can be critical as choice of integration, or aggregation, rule can influence the final IEA outcome profoundly (Ojaveer and Eero, 2011; Borja *et al.*, 2013; Caroni *et al.*, 2013). Borja *et al.* (2014) consider nine possible methods for combining or aggregating the results of individual indicator assessments to derive an overall IEA outcome. In a later section we summarise each option and consider its merits with respect to the IEA necessary to meet MSFD needs.

Management strategy evaluation

Having carried out an IES on the whole suite of indicators and determined the current holistic state of the marine ecosystem, the next step requires the optimal management strategy to be identified, across all the human activities that impact the ecosystem (Sainsbury *et al.*, 2000; Bunnefeld *et al.*, 2011). Management strategy evaluation (MSE) is the term used to describe this process, which usual involves the use of models to determine which combination of management measures, applied across the range of human activities that impact the marine ecosystem, will have the most beneficial outcome on ecosystem state, and is therefore the most likely to achieve the over-arching goals of the EBM policy (Smith *et al.*, 2007; Levin *et al.*, 2009; Heath, 2012; Kaplan *et al.*, 2012; Levin *et al.*, 2014). Models used to support management strategy evaluation have to address at least three sources of uncertainty: process uncertainty (e.g. what influences variation in the biomass of a fish stock), data uncertainty (e.g. how accurately is stock biomass measured), and implementation uncertainty (e.g. how closely are management measures adhered to) (Butterworth and Punt, 1999).

Management strategy evaluation has played an important role in fisheries management (Kraak *et al.*, 2010). MSE has been applied to South African hake stocks to assess the trade-off between stock recovery and yield to determine the optimal recovery strategy with least impact on fisheries incomes (Plagányi *et al.*, 2007). MSE has proved particularly valuable when considering mixed fisheries advice (Mackinson *et al.*, 2009; Ulrich *et al.*, 2011; Rijnsdorp *et al.*, 2012). More recently methods have been developed to extend this process to evaluate the consequences of different fisheries management strategies on other components of marine ecosystems that EBM might be expected to address (Kraak *et al.*, 2012; Gascuel *et al.*, 2012). However, the number of studies that extend management strategy evaluation to include the impacts of other human associated pressures on marine ecosystems beyond the principal fish stocks remains relatively limited, but include activities such as gravel extraction and the building of offshore windfarms (Stelzenműller *et al.*, 2010; Foden *et al.*, 2010; 2011; Stelzenműller *et al.*, 2011).

These studies suggest that it is unlikely that any single combination of management measures will consistently provide an optimal outcome across all management objectives. It seems probable that trade-offs will have to be made; achieving objectives for one ecosystem component, and thereby meeting targets for indicators monitoring change in this component, might only be possible at the cost of failing to meet targets for indicators monitoring change in another ecosystem component (Levin *et al.*, 2009). Management strategy evaluation will also have to take account of the cumulative effects of multiple pressures impacting on each ecosystem component (Stelzenműller *et al.*, 2010; Foden *et al.*, 2010; 2011). Cumulative effects can be additive (combined impact equals the sum of the two individual impacts), synergistic (the effect of one pressure magnifies the impact from another so that the combined impact is greater than the sum of the individual impacts) or antagonistic (the effect of one pressure reduces the impact from another so that the combined impact is less than the sum of the individual impacts) (Crain *et al.*, 2008; Griffith *et al.* 2012; Gobler *et al.* 2014). In the case of synergistic and antagonistic cumulative effects, indicator responses to management measures that reduce one of the interacting pressures are unlikely to follow trajectories anticipated from the pressure-state relationships for the pressure in question because of the interacting effects of the second pressure. It is likely therefore that no single management measure will adequately address the cumulative effects of multiple pressures (Boldt, *et al.*, 2014), appropriate management strategy evaluation will therefore be essential for identifying optimal combinations of measures (Griffith *et al.* 2012).

Monitoring of ecosystem indicators and management effectiveness

Having completed an IEA, used management strategy evaluation to identify the optimal combination of management measures, and then implemented these measures does not bring the exercise to an end. IEA is a cyclical process. Ongoing monitoring of the marine ecosystem using the suite of ecological indicators, with periodic assessments of overall ecosystem state to define how this changes into the future, is necessary not only keep a watch for any future changes in status that might be attributable to new human activity, but also to assess the effectiveness of the management that has been put in place (Walters, 1986; Levin *et al.*, 2009). In many instances such ongoing management is neglected or inadequate and the effectiveness of management therefore unclear (Rumps *et al.*, 2007). This can result in further unexpected degradation of the marine ecosystem, which can then incur unnecessary social and economic costs if the management measures now needed to reverse this are even more stringent and restrictive that would otherwise have been the case (Shertzer and Prager, 2007).

Progress in applying the six step framework within the MSFD

Figure 6 illustrates Levin's *et al.* (2009) cyclical six-step IEA framework outlined above, but with each step in the framework colour coded to indicate current progress in applying the framework to the MSFD. We now elaborate on this to assess progress towards completing an IES that meets MSFD needs.

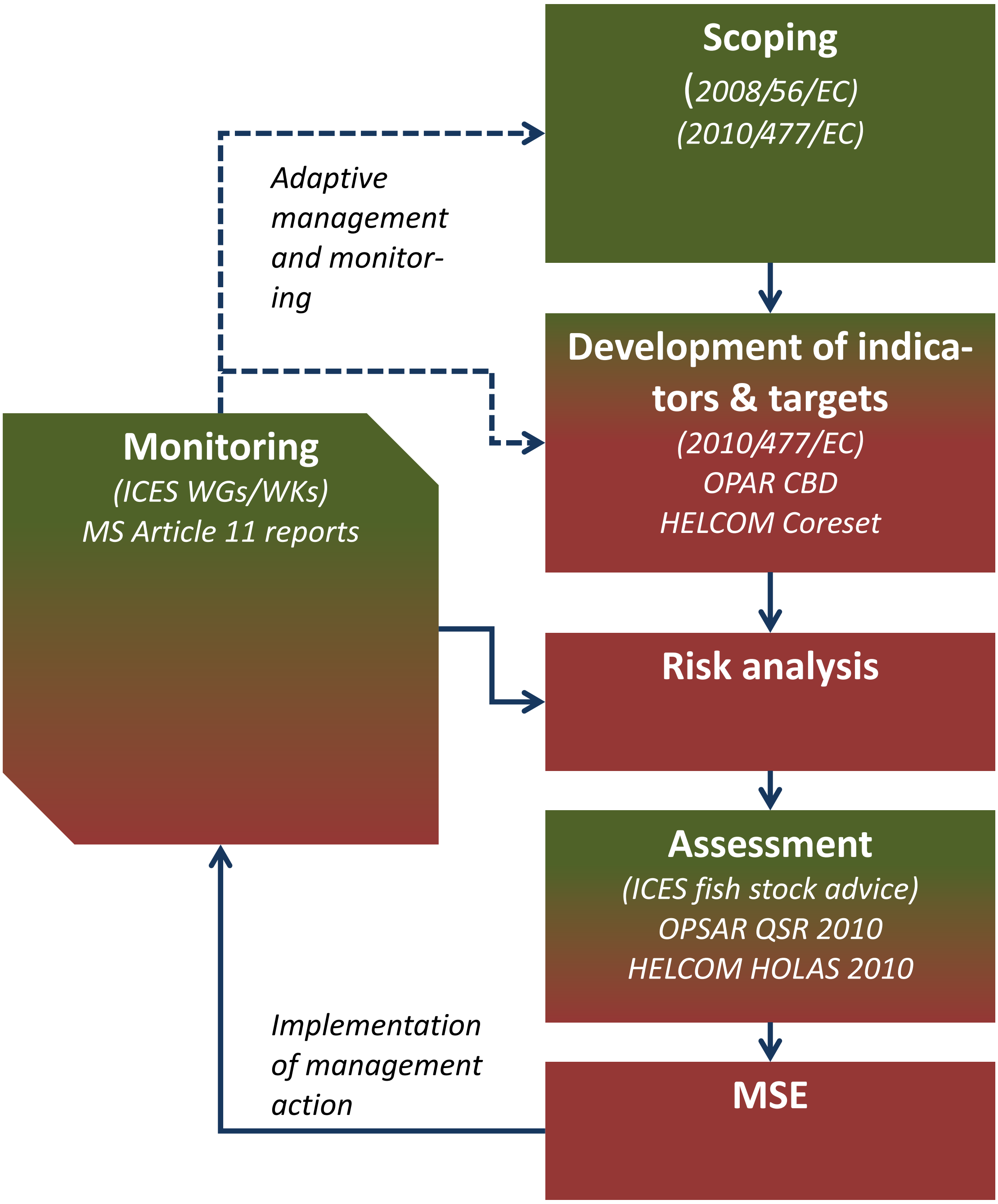


Figure 6. The framework for integrated ecosystem assessments according to Levin *et al.* (2009). Colour shadings indicate the current stage of implementation of such an IEA-framework within the MSFD:

green - already implemented, red - not implemented yet, colour gradient - partly implemented. MSE stands for management strategy evaluation.

Scoping

In implementing the MSFD, scoping has been performed at several levels. The third paragraph of the Directive preamble explicitly expresses its over-arching goals, its vision (e.g. Sainsbury and Sumaila, 2003; Levin *et al.*, 2014), "*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*" (EC, 2008). Article 1 of the MSFD expands further on this: paragraph 1 states "*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*", thereby explicitly stating both the overarching goal and a time-frame. Paragraph 5 of Article 3 defines what is meant by good environmental status (GES) and concludes by stating "*Good environmental status shall be determined at the level of the marine region or subregion as referred to in Article 4, on the basis of the qualitative descriptors in Annex I*". Annex I lists eleven qualitative Descriptors for which GES must be achieved (see Table 2), starting the process of establishing the required operational objectives (e.g. O'Boyle and Jamieson, 2006; Dickey-Collas, 2014). Article 4 explicitly states the spatial scales at which GES needs to be achieved for each Descriptor; the spatial scales relevant to each operational objective.

The spatial scales outlined in the MSFD align closely with areas covered by the Regional Seas Conventions (RSCs); they include the Baltic Sea (covered by HELCOM), the North-east Atlantic Ocean (covered by OSPAR), the Mediterranean Sea and the Black Sea. Two regions are further partitioned into defined subregions. The North-east Atlantic Ocean is split into the Greater North Sea, including the Kattegat and the English Channel, the Celtic Seas, the Bay of Biscay and the Iberian Coast and in the Atlantic Ocean, the Macaronesian biogeographic region, being the waters surrounding the Azores, Madeira and the Canary Islands, and the Mediterranean Sea is subdivided into the Western Mediterranean Sea, the Adriatic Sea, the Ionian Sea and the Central Mediterranean Sea, and the Aegean-Levantine Sea. In the North-east Atlantic, these MSFD subregions also align closely with the OSPAR regions. Article 6 of the MSFD outlines the responsibilities of the RSCs in coordinating monitoring, assessment and management activities at the scale of these regions and subregions defined in the MSFD. The RSCs have also therefore been instrumental in the scoping process, especially with regard to identifying the specific ecosystem components, 'benthic habitats', 'pelagic habitats', 'fish and cephalopods', 'seabirds and reptiles' and 'marine mammals', that should be monitored and assessed, particularly in respect of Descriptors 1 (biological diversity is maintained), 4 (food web structure and function) and 6 (seafloor integrity). Article 8 explicitly requires MSs to identify the essential features and characteristics of their waters based on an indicative list of elements set out in Annex III (see Table 3). The specific key ecosystem components specified by the RSCs take this aspect of the scoping process a step further.

Although overall assessment of the state of marine ecosystems has to be made at the regional or subregional spatial scale, this does not mean that the actual analysis on which such an assessment is made need necessarily by carried at the same spatial scale. Within marine areas the size of the Greater North Sea Subregion, for example, considerable spatial heterogeneity in species composition and relative abundance has been demonstrated among some communities. For example, fish and benthic invertebrate communities in the northern and southern North Sea differ markedly (Basford *et al.*, 1989; 1990; Callaway *et al.*, 2002; Fraser *et al.*, 2008). In such circumstances it might make more sense ecologically to perform separate assessments of the status of these disparate communities. Article 4 of the MSFD permits assessments to be made at spatial scales smaller than that of the subregion, the subdivisional spatial scale, but the outcomes of these subdivisional assessments need subsequently to be integrated in order to draw conclusions regarding overall status of fish and benthic communities across the whole subregion. Greenstreet *et al.* (submitted) suggest that subdivisional assessments of fish communities using the LFI might be more appropriate, thus taking account of the specificities of the fish community in different parts of the North Sea, as well as any geopolitical partitioning in order to meet individual MSs reporting needs (Figure 7). Integration of the analysis outcomes would then allow conclusions to be drawn regarding the status of the fish community North Sea wide. Conversely, biomass indices for a 'suite of sensitive species' relate to the abundance of the whole population occupying the North Sea (Greenstreet *et al.*, 2012). Analysis of these indicators is therefore more appropriately carried out across the whole subregion.

Figure 8 presents a second example where a strong case for subdivisional scale assessment could be made; this time for subdivisions with a Region. In the Baltic Sea, the huge gradient in salinity, and all the ecological variation associated with this, would suggest that single assessments made at the scale of the whole Region could mask variability that would be apparent should assessment be made at an appropriate subdivisional scale.

In some Regions/Subregions, monitoring programmes do not cover the entire area, hence assessments would most likely be carried out using the individual survey monitoring programmes. In both instances, in coming to an overall assessment of status for a particular indicator at the Subregional scale, integration of the individual sub-divisional scale assessments would be required.

Identifying a key threats to the marine ecosystem is another key aspect of scoping prior to any IEA and again Article 8 of the MSFD requires MSs to undertake "*an analysis of the predominant pressures and impacts, including human activity, on the environmental status*" of their waters (see Table 4). Once again the RSCs have been instrumental in prioritising the key pressures impacting marine ecosystems across the regional sea. Table 5 provides an overview of the scoping across both ecosystem components and pressures by both MS meeting their MSFD obligations and by the RSCs.

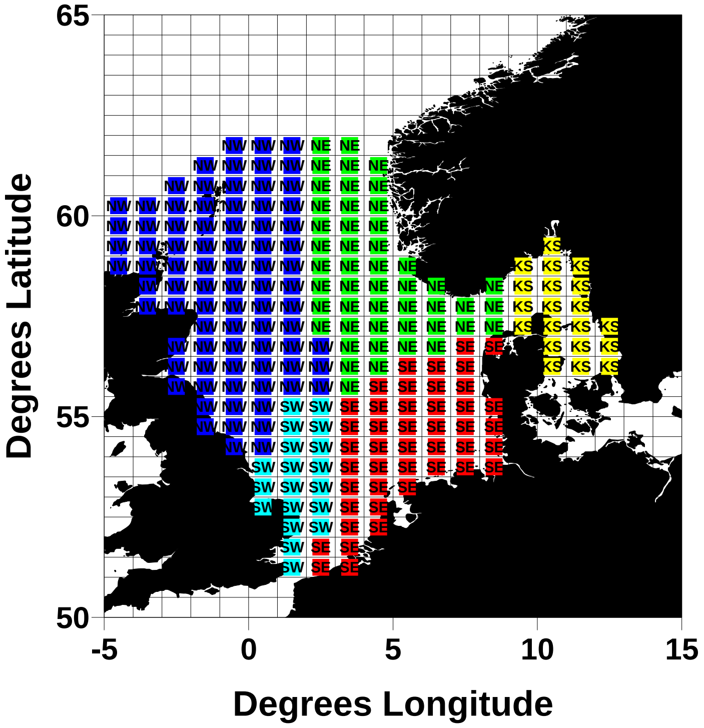


Figure 7. Partitioning of the Greater North Sea Subregion into five spatial subdivisions reflecting both biological and geopolitical concerns (NW northwest North Sea; NE northeast North Sea; SW southwest North Sea; SE southeast North Sea; KS Kattegat-Skagerrak).

Figure 8. Map of the Baltic Sea presenting sub-divisions into 17 open sub-basins and 42 coastal areas used for the HOLAS assessment.

Table 5 Overview on scoping by the EU-Commission (2008), OPSAR (2010) and HELCOM (2009) on relevant biodiversity components and anthropogenic pressures to be included (X) within the assessment of environmental status.

Ecosystem components	HELCOM	OSPAR	MSFD (Annex III, Tables 1&2)
Species			
Fish	X	X	X
Seabirds	X	X	X
Marine mammals	X	X	X
Benthic invertebrates	X	X	X
Reptiles	O	X	X
Macro-algae	X	X	X
Habitats			
Benthic	X	X	X
Pelagic	O	X	X
Communities			
Fish	X	X	X
Phytoplankton	X	X	X
Zooplankton	X	X	X
Benthic invertebrates	X	X	X
Food webs	O	X	
Non-indigenous species	X	X	X
Anthropogenic pressures			
Marine resources			
Fishing	X	X	X
Mariculture	O	X	X
Hunting	X	X	X
Wind Farms	X	X	X
Oil & gas extraction	X	X	X
Aggregate extraction	X	X	X
Traffic and infrastructure			
Shipping	X	X	X
Tourism	X	X	X
Cables	X	X	X
Coastal defence	X	X	X
Dredging & dumping	X	X	X
Pollution & Contamination			
Eutrophication	X	X	X

Hazardous substances	X	X	X
Radioactive substances	X	X	X
Underwater noise	X	X	X
Marine litter	X	X	X
Abiotic & biotic disturbances			
Climate change	X	X	O
Non-indigenous species	X	X	X

Scoping was progressed further by ten joint JRC/ICES task groups, each tasked with considering the key features for one designated Descriptor and how these should be monitored to support formal assessment. There was no task group for Descriptor 7. Each task group produced a comprehensive report and these were summarised by Cardoso *et al.* (2010). This ultimately led to the production of Decision document, which for each Descriptor identifies the technical features (termed Criteria) that define GES and the types of indicator that MSs should use to monitor change in these criteria and so assess the status of each Descriptor (EC, 2010). The Decision document therefore provides a clear steer to both the MSs and the RSCs as to the types of metric they need to use to monitor change and assess status of the key ecosystem components in respect of each Descriptor.

In many respects the scoping phase of an IEA to meet MSFD needs might be deemed to be complete (Figure 6), but in one important respect this is not the case. To date little thought has been directed towards deciding how precisely all the information conveyed by the numerous ecological indicators used in monitoring status across the various ecosystem components, Criteria and Descriptors is going to be integrated to arrive at an overall holistic assessment of state of the ecosystem. It is intended that this document should start the process of addressing this omission, and to this end we address this issue directly in later sections. In this respect, we directly address ToR 1d.

Development of ecosystem indicators and targets

As stated above, a major part of the scoping phase has been the identification by the joint JRC/ICES Task Groups of the key elements and attributes of each Descriptor that require monitoring and assessment (Cardoso *et al.*, 2010), ultimately leading to the production of the EC Decision document (EC, 2010). This defined the types of indicators that MSs and RSC would need to base their assessments on. MSs and the RSCs then just needed to identify the most effective metrics to fulfil these indicator functions. MSs used a wide range of different indicators in performing initial assessments of their own waters under article 8, but this use of such a diversity of different indicators by different MSs bordering particular defined MSFD subregions would have rendered IEA impossibly difficult. Following this initial assessment phase therefore, the RSCs, primarily responsible for assessment at the spatial scale of the defined regions and subregions, have focused on distilling a few key, high performing indicators that should be used by all MSs bordering any given region or subregion HELCOM, 2013; OSPAR, 2013a; OSPAR, 2013b). The application of criteria to assess the performance of different indicators has been a key part of indicator selection. Chapter xxx of this report describes such a process that has been used in the selection of OSPAR 'common indicators'.

OSPAR calls indicators thus selected 'common indicators', several of which are now considered to be fully developed; the metric in question can be supported by ongoing monitoring programmes and metric target values can be set so that it can support formal assessment. Examples of such metrics for the fish and cephalopod ecosystem component under Descriptor 1 include the Large Fish Indicator which has been shown to be operational across most of the Greater North Sea subregion, and in spatial subdivisions of the Celtic Seas and Bay of Biscay and the Iberian Coast subregions (Greenstreet *et al.*, 2011; Shepherd *et al.*, 2011b; Modica *et al.*, 2014), and the biomass indices for a suite of 'sensitive' species which has been demonstrated in the Greater North Sea (Greenstreet *et al.*, 2012). Such metrics have been agreed by all MSs bordering some subregions and are therefore considered to be fully operational 'common indicators'. Several more metrics have yet to reach this level of development and are therefore listed as 'candidate indicators' with specific MSs nominated to progress their development. The current intention is that assessment of the state of the marine ecosystem will be assessed at the spatial scale of the defined regions and subregions using these 'common indicators' and perhaps a select few 'candidate indicators' if these can be developed sufficiently in the time remaining before the results of the assessment need to be reported in 2017.

Risk analysis

Paragraph 6 of Part A of the Decision document states "*A combined assessment of the scale, distribution and intensity of the pressures and the extent, vulnerability and resilience of the different ecosystem components including where possible their mapping, allows the identification of areas where marine ecosystems have or may have been adversely affected. It is also a useful basis to assess the scale of the actual or potential impacts marine ecosystems. This approach, which takes into account risk-based considerations, also supports the selection of the most appropriate indicators related to the criteria for assessment of progress towards good environmental status.*" This statement alludes to exactly the sort of risk analysis that many authors have suggested is needed to identify the critical indicators, or components, that need to be given greatest weight in an IEA in coming to a conclusion regarding overall status of the marine ecosystem (see above). As yet, little in the way of formal risk assessment has been carried out in support of implementation of the MSFD. Some cross-tabulation of pressures by ecosystem component has been undertaken on a region by region basis (Connor *et al.*, 2009; Breen *et al.*, 2012), which subsequently contributed to the OSPAR Quality Status Report (QSR) 2010 (OSPAR, 2010). Some MSs have started the process of mapping pressures on particular ecosystem components, or by Descriptor, within particular marine regions (Breen and Murray, 2014; Church *et al.*, 2014; Eassom *et al.*, 2014; Jenkins, 2014). Korpinen *et al.* (2012) provide an example of such a mapping of pressures and components approach for the Baltic Sea (Figure 9).

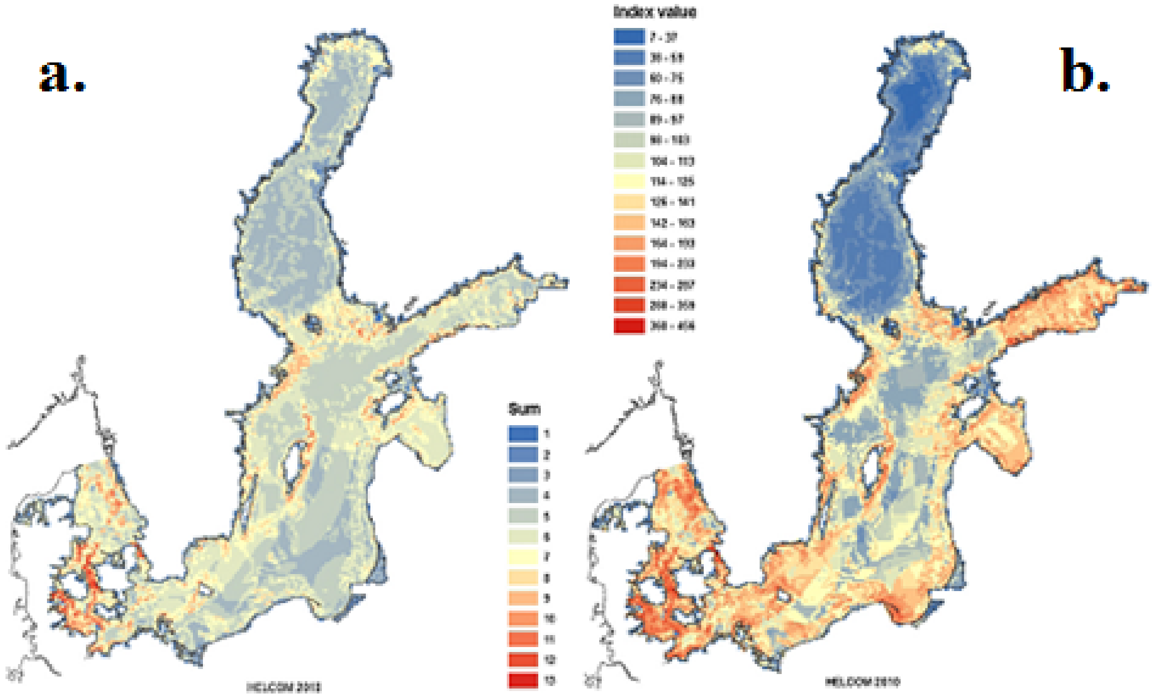


Figure 9. Overlaid presence of 14 ecosystem components of biodiversity (a.) and potential cumulative anthropogenic impacts (b.) in the Baltic Sea. Spatial resolution is 5x5 km. From Korpinen *et al.* (2012).

Although not the result of the sort of explicit risk assessment analysis suggested by Samhouri and Levin (2012), several of the OSPAR 'common indicators' have been selected on the implicit assumption that they represent a 'high-risk' situation; location D in the two-dimensional sensitivity-exposure space plot, (Figure 5). For example the LFI was explicitly 'designed' to maximize its sensitivity to fishing in a marine region where fishing was likely to have had a marked impact of fish community size composition (Greenstreet *et al.*, 2011). Similarly, certain life history traits, such as large ultimate body size, slow growth rate, late age and large size at maturity, etc, are known to render fish species characterized by these traits especially sensitive to fishing mortality (Jennings *et al.*, 1998; Gislason *et al.*, 2008; Le Quesne and Jennings, 2012) and populations of species with these life history traits have declined markedly in many marine regions where fishing has been extensive (Greenstreet and Hall, 1996; Rijnsdorp, *et al.* 1996; Philippart, 1998; Walker and Hislop 1998; Greenstreet *et al.*, 1999; Frisk *et al.*, 2001; van Strien *et al.*, 2009). The 27 species chosen to populate the 'population biomass of a suite of sensitive species' indicator were all selected on the basis of having life history traits likely to render them the most susceptible to fishing pressure in a marine region where such species are known to have been subjected to heavy and prolonged fishing mortality (Greenstreet *et al.*, 2012). Although no formal risk assessment has been carried, there can be little doubt that this indicator again monitors a 'high-risk' situation.

Assessment of ecosystem status

Under article 8 of the MSFD, MSs were obliged carry out an initial assessment of the status of the marine ecosystem in their own exclusive economic zones (EEZs) and report by June 2012. This was largely done independently by the individual MSs; each MS applying their own suite of selected indicators to their own defined set of ecosystem components. Little attempt was made to coordinate the use of particular indicators across shared marine regions. Across EEZs within the MSFD regions and subregions there was little standardization in the approaches adopted to integrate the information conveyed by these various suites of indicators to derive overall holistic assessments of status at the EEZ scale. Consequently, it was not feasible to assess overall status at the scale of the region and subregion from the various MS EEZ initial assessments. However, in line with paragraph 24 of the MSFD preamble, MSs within the OSPAR area, for example, might have presumed that the OSPAR QSR 2010 would have provided the necessary regional- and subregional-scale assessment (OSPAR, 2010).

Article 17 requires MSs to update their assessments on a six-year cycle. Since the initial assessment was required by July 2012, the next assessment is due in July 2018. The next OSPAR interim assessment is scheduled for 2017 (IA2017). This is expected to meet MSFD July 2018 assessment needs at the regional and subregional spatial scale. The assessment phase of the six-step IEA framework is therefore imminent and the work required to meet this assessment schedule should already have commenced and be ongoing. The need for this is also explicitly stipulated in Article 5 of the MSFD requiring MSs to have adequate monitoring programmes in place to support ongoing assessment needs by July 2014.

Management strategy evaluation

Article 8 requires MSs to undertake "*an economic and social analysis of the use of those waters and of the cost of degradation of the marine environment*" implying the need for some form of MSE (EC, 2008), but to date, examples of MSE explicitly to support implementation of the MSFD are unknown. It is perhaps too early for MSFD implementation to have reached this stage of the six-step IEA framework. Article 5 of the MSFD only requires MSs identify the management measures necessary to achieve GES by July 2015 and to implement these measures by July 2016. As stated in the section above, the next assessment is not due until July 2018, or June 2017 for the OSPAR IA2017. Perhaps only when the current status of different components of the marine ecosystem has been assessed, and the management measures necessary to meet the various indicator targets identified will the need for proper MSE be fully appreciated.

It is likely that two issues in particular are going to arise during the upcoming IEA that will require MSE to fulfil the OSPAR IA2017 and MSFD July 2018 assessment needs. Firstly during the indicator selection and target setting phase of the IEA six-step framework described above, the majority of indicators and targets have been considered in isolation; little consideration has been given to the types of trade-off that will almost certainly be necessary to resolve potential incompatibilities between some indicator targets (Sissenwine and Murawski, 2004; Link, 2010). For example, the general perception is that marine regions such as the North Sea have been overfished for the much of the latter half of the 20th century (Thurstan *et al.*, 2010), so that many stocks have been depleted

(Piet and Rice, 2004; Probst *et al.*, 2013) and leaving the fish community in a perturbed state (Greenstreet and Rogers, 2006), and in particular with a much reduced abundance of higher trophic level piscivorous fish (Christensen *et al.*, 2003; Myers and Worm, 2003). Targets for fish stocks and fish community indicators are therefore set in line with instigating recovery towards some former historic state. Conversely, many seabird populations increased in size over most of the 20th century and targets for many seabird indicators aim at maintaining a more contempory state. Because of the different reference periods being implicitly adopted, these targets may be incompatible. If this proves to be the case then MSE may be necessary to resolve the situation. The cumulative effects of multiple human pressures on state indicator values is the second major outstanding issue that will also need to be addressed if the results of the upcoming IEA to meet OSPAR and MSFD needs are to be successfully interpreted to formulate appropriate management advice, and again MSE is likely to produce the optimal outcome in this respect.

Monitoring of ecosystem indictors and management effectiveness

Since Article 5 of the MSFD only requires MSs to have implemented the management measures necessary to achieve GES by July 2016, the next assessment phase, two years later in 2018, will provide the first opportunity of gauging the efficacy of these management measures. This ongoing monitoring ecosystem change and management effectiveness phase of the six-stage IEA framework will therefore only really commence following the OSPAR IA2017 and MSFD July 2018 assessment.

Indicator-based management frameworks as the basis for IEA

Indicator-based management frameworks were devised several decades before the introduction of the MSFD. In the beginning of the 1990s the Organization for Economic Cooperation and Development (OECD) proposed the use of the Pressure-State-Response framework which linked three specific types of indicator in a formalized framework to support environmental assessment and management (OECD, 1993). Historically, the capacity effectively to populate indicator management frameworks, such as the PSR framework (Garcia and Staples, 2000), has primarily influenced ICES earlier advice regarding the selection of operational “state” indicators (Greenstreet 2008). Single species focused fisheries management has traditionally been implemented on the basis of a PSR indicator-based management framework; the *pressure* indicator, fishing mortality (F), has been manipulated by altering the *response* indicator, the total allowable catch (TAC), in order to achieve management objectives for the *state* indicator, spawning stock biomass (SSB) (Jennings, 2005; Piet *et al.*, 2010). However, in adapting the PSR framework to broaden its application to cover an increasingly diverse range of anthropogenic pressures affecting most major components of marine ecosystems, its shortcomings quickly became apparent. Chief amongst these was the realization that the management *response* rarely influences *pressure* directly. Instead it is the human *activity* that is directly affected by the management *response*, and it is through modifying the *activity* that management can influence *pressure* on an ecosystem component and so alter its *state*.

Within a fisheries management context, annual landings provides the relevant indicator of *activity*; altering TACs directly limited the quantity of fish that fishermen could catch, which in turn indirectly influenced fishing mortality rates. But it was when using these frameworks within an EBFM context that the need for explicit inclusion of indicators of *activity* became apparent. In compliance with the Common Fisheries Policy, quantities of fish landed are routinely recorded and data are collected for the specific purpose of estimating rates of fishing mortality, but similar data are rarely available for non-targeted fish species and benthic invertebrate organisms taken up or damaged by fishing gears. Instead estimates of fishing mortality rates, still the most appropriate indicator of the *pressure* imposed by fishing on these species, have to be estimated from models driven by fishing effort data (Piet *et al.*, 2007; Piet et at., 2009; Fock, 2011; Fock *et al.*, 2011). Thus indicators of fishing effort, an alternative measure of the level of *activity* in the fisheries, are explicitly required in indicator-based management frameworks intended to support EBFM. The traditional fisheries management *response* measure, catch limitation through setting TACs, has proved relatively ineffective means of achieving the desired manipulation of fishing *activity* in terms of time spent fishing (Piet and Rice, 2004; Greenstreet *et al.*, 2009; Reiss *et al.*, 2010a; Piet *et al.*, 2010). Only when specific measures to limit days allowed at sea have been applied have levels of fishing activity (hours fishing) actually declined (Greenstreet *et al.*, 2009). EBFM therefore not only needs an indicator-based management framework that explicitly includes indicators of the level of anthropogenic *activity*, but the actual metric used may well differ from the metric used to support traditional fisheries management, and as a consequence alternative *response* indicators may also be needed. Such considerations have led to development of the PRS framework to the Activity-Pressure-State-Response (APSR) (Greenstreet *et al.*, 2009) (Figure 10).

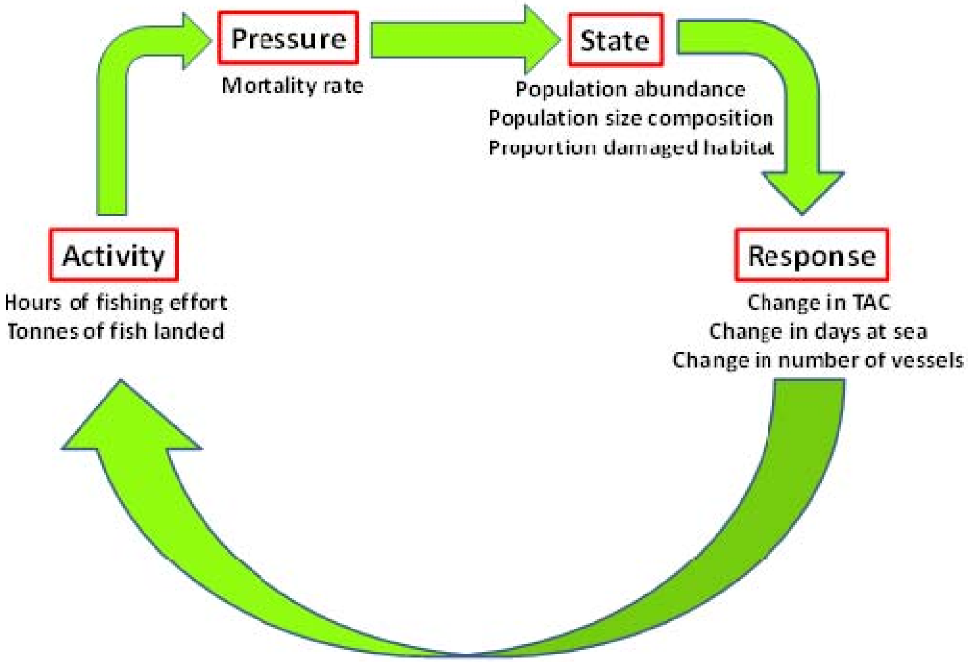


Figure 10. Illustration of the Activity – Pressure – State – Response (APSR) indicator based management framework with some examples of the types of indicator that might be used in each role within a typical fisheries impact situation.

Within the wider marine environment arena, similar developments in the types of indicator-based frameworks used to support management decision-making have also taken place. The PSR framework has been extended by the European Environmental Agency (EEA) promoting a Driver-Pressure-State-Impact-Response (DPSIR) framework (Smith and Weterings, 1999; Cormier *et al.*, 2013). *Driver* indicators represent social and economic driving forces, which promote and influence human *activities* in the marine environment, which in turn exert *pressures* on components of the ecosystem. These *pressures* cause a change in the *state* of these ecosystem components, deemed to be the *impact*, and this requires a societal or policy *response* implemented by the responsible competent management authorities (Smith and Weterings, 1999). In treating the *driving* forces of the DPSIR framework as the societal forces that influence anthropogenic *activity* in the marine environment, the DPSIR and APSR frameworks do not actually differ that much from one another. To date, PSR or DPSIR frameworks have been employed in various studies (Borja *et al.*, 2006; Gimpel *et al.*, 2013; Smith *et al.* 2014) and have been investigated by various scientific disciplines (Luiten, 1999; Elliot, 2002; Svarstad *et al.*, 2008; Maxim *et al.*, 2009). Further developments of the DPSIR framework have also been reported, such as the DPSWR framework in which *impact* has been substituted by human *welfare* (Cooper *et al.* 2013) and the enhanced DPSIR (eDPSIR), which is based on causal networks (Niemeijer and de Groot, 2008). Consideration has also been given as to how the DPSIR framework might be used to support implementation of the MSFD (Knights *et al.*, 2013).

Links in the APSR indicator-based framework

A major advantage of indicator-based management frameworks is that key relationships are captured and displayed in a relatively simple way, helping to structure and focus the entities requiring consideration and management. Even when the correct indicators have been identified and linked together in an APSR (or equivalent) management framework, the precise formulation of the individual metrics performing each indicator role can often be modified to increase their performance within these roles. Thus the Large Fish Indicator (LFI), which supports the North Sea EcoQO for fish communities, underwent several transitions in its precise calculation with the express purpose of increasing its sensitivity to variation in fishing mortality (i.e. to increase the strength of the linkage between "pressure" and "state") and reducing its sensitivity to other drivers, such as environmentally related variation in recruitment (Greenstreet *et al.*, 2011). Then, in developing the LFI for application in other marine regions, such as the Celtic Sea or southern Bay of Biscay, formulation of the LFI underwent further revision to derive the most effective indicator in each new area (Shephard *et al.* 2011; Modica *et al.*, 2014). Similarly, Piet *et al.* (2007) describe the process of developing and combining different types of information related to the process of fishing to derive increasingly effective indicators of fishing "activity" and "pressure". Greenstreet *et al.* (2009) exemplify this process, incorporating information on the number of vessels in different metiers of the Scottish fishing fleet, their engine power, the duration of the individual trips they make, the distance travelled in each trip, and the time involved in carrying out different parts of the fishing exercise to derive the most precise estimates possible of annual fishing effort in each ICES statistical rectangle in each year across the North Sea. It is precisely this sort of detailed "activity" information that models, such as the one developed by Piet *et al.* (2009), require to determine reliable estimates of fishing "pressure" on components of the marine ecosystem not directly targeted by fisheries, and so not explicitly monitored to support fisheries management. In this case the model algorithm itself describes a fairly complex relationship between fishing "activity" and fishing "pressure".

Each link in the A-P-S-R chain is now considered in detail.

Pressure–State

The nature of the pressure state relationship can vary considerably. For example the "directness" of the relationship can differ markedly in different situations. The response of the LFI to variation in fishing mortality tends to be lagged with lags of 15 years reported in the North Sea (Greenstreet *et al.*, 2011), 10 to 12 years in the Celtic Sea (Shepherd *et al.*, 2011b), and only 6 years in the Bay of Biscay (Modica *et al.*, in press). Lag times in the response in response to changes in fishing pressure may also vary between species (Probst *et al.*, 2012). The pressure state relationship might also differ between disturbance and recovery phases. Again using the LFI as an example, the lagged relationship between the LFI and fishing mortality seems to be progressively weakening as the lag period increasingly includes more years of declining fishing mortality and fewer years of increasing mortality (Fung *et al.*, 2012). Invariably conditions differ between the recovery and disturbance phases, and this can alter the nature of the pressure-state relationship:

Loss of large predatory fish such as cod can lead to reduced predation pressure on their fish prey, sandeels, herring and sprat. Populations of these pelagic species might therefore increase, leading to increased predation pressure on the cod eggs and larvae (Daan *et al.*, 1985; Köster and Möllman, 2000. Cod recruitment potential in the recovery phase might therefore be lower than during the disturbance phase. Fishing removes physical habitat structures that provide shelter to settling juvenile demersal fish (Auster *et al.*, 1996; NOAA 1998; Auster and Langton, 1999; Lindholm *et al.*, 1999; Turner *et al.*, 1999; Wilson *et al.*, 2010). During the recovery phase therefore, when these refuges have been lost, juvenile demersal fish may be subject to higher natural mortality rates through increased predation. Fish community trophic structure may have been maintained despite changes in species and size composition. Large piscivorous fish may have been removed by fishing, but replaced by smaller piscivore species. With a reduction in fishing mortality, the large predator species might be expect to increase in abundance once again, but instead they now face competition for these smaller piscivores (Jennings *et al.*, 2002). The size–composition recovery response to changing fishing mortality might well be much slower than the disturbance response.

Changing environmental conditions might well alter conditions in such a way as to affect the relationship between pressure and state. Industrial fisheries had a negative impact on kittiwake breeding success. But in recent years, warming seas have affected plankton communities and dynamics, altering the food supply to sandeels, a key prey species for provisioning kittiwake chicks. Kittiwake chick productivity is now reduced, even in the absence of industrial fishing (Frederiksen *et al.*, 2004).

Activity–Pressure

Again the nature of the relationship needs to be considered; the relationship could be linear, such that change in pressure is directly proportional to variation in activity, or it can be non-linear. For example, benthic invertebrate mortality does not increase linearly with increasing fishing effort, instead the rate of increase in mortality falls off rapidly as effort increases (Greenstreet *et al.*, 2009). This is because fishing effort is patchily distributed (Piet *et al.*, 2007; Piet and Quirijns, 2009; Lambert *et al.*, 2012). Fishermen return to the same locations and fish them repeatedly, but benthic invertebrates can only be killed once! Most of the damage occurs in the first fishing event and with each successive fishing event, the actual number of invertebrates lost declines. Activity-Pressure relationships vary markedly between species; some species are more sensitive than others. They can also be influenced by environmental conditions; the susceptibility of a benthic invertebrate to fishing impact is influenced by the physical nature of the seabed (Collie *et al.*, 2000; Kaiser *et al.*, 2006). The relationship might also differ between different activities; for example one might monitor the relationship between the mortality of a benthic invertebrate species and the hours of fishing effort by otter trawlers and beam trawlers – in each case the indicators are ostensibly the same, but the activity-pressure relationships for the two fishing metiers will probably be very different (Kaiser *et al.*, 2006).

Response-activity

This linkage is not actually explicit in the APSR chain, but it in fact closes the loop and is perhaps the most critical interaction of all within the APSR indicator framework. Unless this relationship is well understood, and formally considered for each and every management measure, than it is very unlikely that the intended goals of management will actually be achieved (Piet *et al.*, 2010).

The first thing to consider here is that this link in the indicator framework involves the interaction between humans; between the managers of an activity and the participants in an activity. It can therefore be highly contrary. For example, mortality of sensitive non-target fish species is related to variation in fishing effort (Piet *et al.*, 2009), so when direct restriction of fishing effort was introduced, leading to a 40% reduction in fishing effort, it might have been anticipated that such species might well have benefitted from this management measure. However, fishing effort decreased most in the most heavily fished parts of the North Sea, and actually increased slightly in previously unfished or very lightly fished areas (Greenstreet *et al.*, 2009). These increases in effort in areas where fishing effort was previously zero probably caused harm that outweighed any benefit gained from the decline in fishing effort in heavily fished area. Fishing effort in these heavily fished areas still remained too high to allow recovery of sensitive fish species.

Similarly, when landings fulfilled the main fishing “activity” indicator role, TACs were the principal “response” indicator used to control the fishing “pressure” indicator, fishing mortality on the targeted stocks. Reducing TACs may have reduced fishing pressure on commercial stocks, but this management response would have been ineffective in reducing the impact of fishing on non-target fish species. Such species are primarily affected by variation in fishing effort, and changing TACs alone had negligible impact on fishing effort levels. Fishermen changed their behaviour, essentially by fishing less efficiently, so that although landings declined in line TACs, thereby reducing fishing mortality on targeted stocks, they actually expended the same amount of effort to take these smaller catches (Reiss *et al.*, 2010a). The impact of fishing on non-target species and benthic invertebrates remained the same, or perhaps even higher if this change in behaviour took their activity into previously unfished locations.

Because of the contrary nature of humans, management measures intended to achieve specific goals can be nullified, or have other unintended consequences, because the nature of the activity changes in other respects that were not anticipated.

State-Response

At the initiation of management to mitigate the undesirable impacts of a human activity on the state of a particular ecosystem component, the deviation between the actual state of the component and a state that is deemed acceptable profoundly steers the management response; the amount of recovery required dictates the severity of the management restriction placed on activity. But the importance of this link in the chain throughout the recovery process must also be appreciated. As the state of the ecosystem responds to all the subsequent changes taking place along the A-P-S-R chain initiated by the first management action, managers need also to keep responding by modifying or tempering their management measures.

Indicator-based management frameworks and MSFD implementation

Several authors, as well as agencies such as the European Environmental Agency (EEA), have suggested that indicator-based management frameworks should provide the platform for environmental assessments (Smith and Weterings, 1999; Jennings, 2005; Niemeijer and de Groot, 2008). The use of PSR-indicator clusters would in fact enhance the current structure of the MSFD, in which currently only pressure and state indicators are specified and the relationships between these are often only implied rather than explicit. A PSR or DPSIR indicator-based framework would ensure higher congruency with the MSFD-reporting structure than the current pressure state-relationships (Figure 11). Perhaps future revisions of the Commission Decision (EC, 2010) should include a requirement for assigning recommended indicators to particular indicator categories (e.g. driver, pressure, state, impact or response).

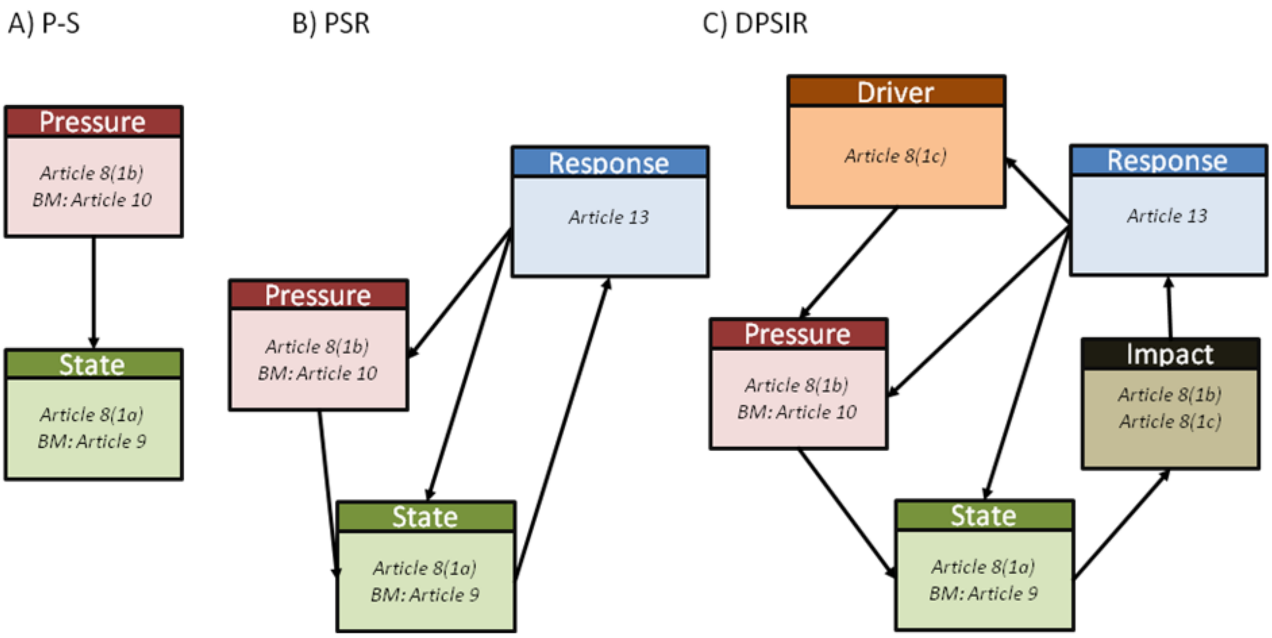


Figure 11. Indicator categories and their relationship to MSFD reporting obligations of Articles 8(1a) (Current status), 8(1b) (Pressures and impacts), 8(1c) (Socio-economic impacts), 9 (Description of GES), 10 (Environmental targets) and 13 (Program of measures). A) Pressure-state relationships as currently implemented within the MSFD. B) Pressure-state-response indicator frameworks (PSR). C) Driver-pressure-state-impact-response frameworks (DPSIR). While P-S will not inform on indicators for Article 10 and Article 13, PSR will inform on all but Article 8(1c), which could be addressed by driver- and impact-indicators. BM: benchmarks of state and pressure indicators could be used for reporting on Article 9 and Article 10, respectively.

To our knowledge, however, few if any of these indicator-based management frameworks have been used explicitly to support existing environmental marine policies. One disadvantage with these frameworks currently is that linkages tend to be viewed independently of each other: without consideration of the interplay with linkages arising from other sectors, the range of pressures generated by specific activities, or the variety of ecosystem components that are impacted by a particular pressure, an issue to which we return in following sections. This may explain their limited take-up as moves to adopt EBFM and EBM have gathered momentum (Knights *et al.*, 2013). Recent developments in the DPSIR framework, the eDPSIR, may start to address these shortcomings. For example, Niemeijer and de Groot (2008) propose the use of ‘causal networks’ based on the eDPSIR framework and built around ‘domains of interest’, for example the impact of fishing on biodiversity (Figure 12). In simple causal networks the number of connections between individual elements serves to identify the key indicators within each component. It is conceptually feasible that causal networks could be constructed for each Descriptor or Criterion in the MSFD Decision document (EC, 2010).

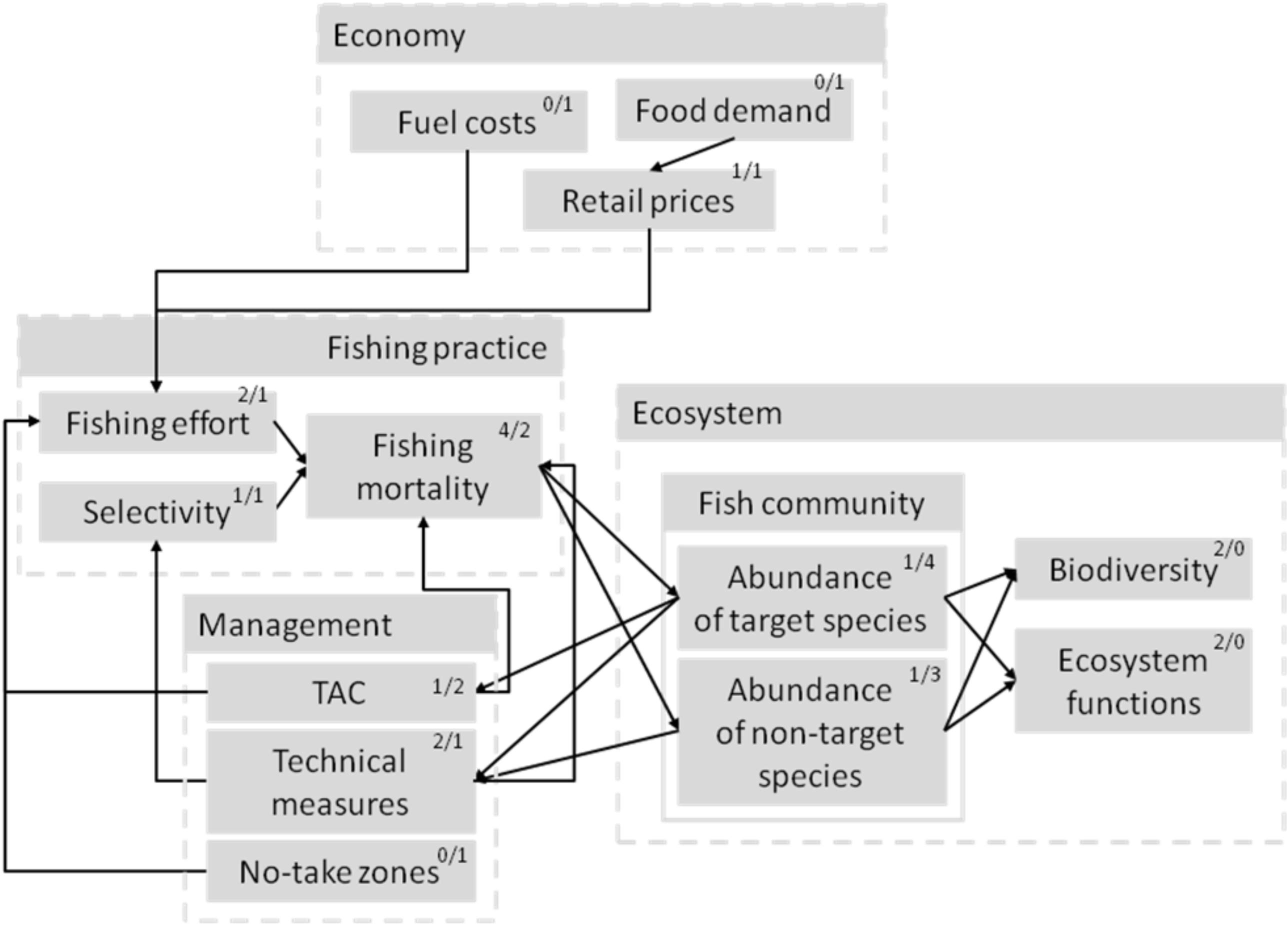


Figure 12. Example of a causal network addressing the impact of fishing on biodiversity. Different components are bounded by boxes with dashed lines. The numbers in the upper right corner of each indicator refers to the number of ingoing and outgoing connections. Note that this is an exemplary case study only intended to demonstrate the structure of causal chain networks.

Outstanding issues affecting successful IEA under the MSFD

Thus far, three major outstanding issues have been identified that must be addressed if the upcoming IEA to meet OSPAR IA2017 and MSFD July 2018 needs is to be successfully concluded. These are:

1) The integration procedures that should be adopted to aggregate the large quantity of information conveyed by numerous indicators used to monitor and assess the status of various attributes of different ecosystem components and pressures under all eleven Descriptors (WGBIODIV ToR 1d);

2) The consequences of multiple human pressures on ecosystem components and accounting for interaction between pressures causing different cumulative effects (WGBIODIV ToR 4);

3) Potential incompatibilities between the targets set for different indicators, particularly indicators that monitor different ecosystem components and the extent and effects of different human pressures on the marine ecosystem (WGBIODIV ToR1b).

In this section we address these three issues in turn.

Integrating the information from multiple indicators

Scoping the integration exercise

The integration exercise needed to perform an IEA to meet MSFD needs must first be scoped out. Paragraph 5 of Article 3 the MSFD document, which states "*Good environmental status shall be determined at the level of the marine region or subregion as referred to in Article 4, on the basis of the qualitative descriptors in Annex I*" (EC, 2008), provides a good starting point. For example this statement makes it clear that we need only assess status at the level of the region, or if this is subdivided, at the level of the subregions specified in Article 4. The Northeast Atlantic region is subdivided into four subregions; the Greater North Sea, the Celtic Seas, the Bay of Biscay and Iberian Coast and the waters surrounding the Azores, Madeira and the Canary Islands. This statement can be interpreted as implying that status of the marine ecosystem in each sub region need only be assessed; there is no requirement to integrate across subregions to arrive at an assessment for the entire Northeast Atlantic. On the other hand, the Baltic Sea region is not subdivided, hence that assessment must be made at the scale of the whole region, the entire Baltic Sea.

Although the MSFD requires GES to be assessed and reported at the spatial scale of the regions or subregions, it does not stipulate that this is the scale at which the analysis must be made. In fact Paragraph 2 of Article 4 explicitly allows for the opposite, stating "*Member States may, in order to take into account the specificities of a particular area, implement this Directive by reference to subdivisions at the appropriate level of the marine waters referred to in paragraph 1, provided that such subdivisions are delimited in a manner compatible with the following marine subregions*". The marine waters referred to in paragraph 1 are the MSFD regions, while the final phrase refers to the MSFD subregions. This statement allows analysis to be carried out at spatial scales that are smaller than the subregions if there is a sound ecological basis for doing so. Where such a case can be made, ignoring this and carrying out the analytical assessment at the scale of the full subregion potentially risks obscuring the information that scientists need to provide the most appropriate advice to managers (Cardoso *et al.*, 2010). A recent study of the LFI in the North Sea came to exactly these conclusions. Reflecting spatial heterogeneity in the North Sea demersal fish community (Fraser *et al.*, 2008), the full subregional scale LFI trend was only representative of changes taking place in the northern North Sea, not across the whole North Sea. Advice based on the full regional scale LFI trend might have suggested that no further management action was necessary; that the LFI was recovering following measures already implemented. However, LFI analyses at the subdivisional scale suggested that further measures might be required in the southern North Sea (ICES 2014; Greenstreet *et al.*, submitted). In some instances therefore, indicators might need to be analysed at a subdivisonal scale with integration then required to derive an assessment outcome at the subregional scale. For other indicators, analysis might be more appropriately done at the scale of the full subregion, for example the abundance/biomass of a suite of sensitive fish species, where the metric concerned refers to the abundance of the whole population within the subregion.

Annex 1 lists the eleven qualitative Descriptors of GES (Table 2), but in this respect, the MSFD is not definitive. The statement quoted at the head of this section can be interpreted in two ways; either integration is required across all eleven Descriptors to arrive at an overall holistic view of the condition of the marine environment, or this level of integration is not necessary and that status need only be assessed on the basis of each Descriptor individually. Borja (2014) takes the view, for example, that although the MSFD requires status to be assessed on the basis of the eleven qualitative Descriptors, there is nothing in the MSFD that stipulates that a single overall assessment is necessary. The Decision document contributes further confusion in stating "*The criteria for assessing the extent to which good environmental status is being achieved … … in relation to each of the eleven descriptors of good environmental status …*" (EC, 2010), which could be interpreted as implying that GES should be assessed at the level of these criteria. However, this interpretation is not widely held and the general assumption is that integration across the Criteria will be necessary in order to draw conclusions regarding status at the Descriptor level. Further integration across Descriptors may then be required if an overall assessment of status is desired. Borja at al. (2013) have proposed an operational definition of GES: "*GES is achieved when physicochemical (including contaminants, litter and noise) and hydrographical conditions are maintained at a level where the structuring components of the ecosystem are present and functioning, enabling the system to be resistant (ability to withstand stress) and resilient (ability to recover after a stressor) to harmful effects of human pressures/activities/impacts, where they maintain and provide the ecosystem services that deliver societal benefits in a sustainable way (i.e. that pressures associated with uses cumulatively do not hinder the ecosystem components in order to retain their natural diversity, productivity and dynamic ecological processes, and where recovery is rapid and sustained if a use ceases)*''. This could be used as a basis for determining the end point in the integration process, i.e. a final integration step across all eleven Descriptors, or stopping at independent assessments for each individual Descriptor.

The Decision document provides the Criteria for assessing progress towards GES in respect of each Descriptor, and for each Criterion, it suggests at least one type of indicator that should be used to monitor change in status. Ideally the metrics chosen to perform these indicator functions should be quantitative, have associated targets, and should therefore be capable of supporting analytical assessment. Across the eleven Descriptors, the Decision document lists 29 separate Criteria with a total of 56 separate Indicator types

(Table 6). Some Descriptors are relatively Criteria and Indicator 'poor'. For example Descriptor 9 "Contaminants in fish" has a single Criterion with two suggested Indicator types, while Descriptor 11 "Energy introduction" has two Criteria listed, each with only a single suggested Indicator type. For such Descriptors, the level of integration regarded to determine status at the Descriptor level is minimal, presenting a relatively trivial issue to resolve. Conversely, Descriptor 1 "Maintenance of biological diversity" is extremely Criteria and Indicator 'rich', presenting a much more complicated integration problem.

Table 6. Number of Criteria and Indicator types listed in the Decision document for each Qualitative Descriptor of GES (EC, 2010).

Descriptor	Number of Criteria	Number of Indicators
1. Biological diversity	7	14
2. Non-indigenous species	2	3
3. Commercial species	3	8
4. Food webs	3	3
5. Eutrophication	3	8
6. Seafloor integrity	2	6
7. Hydrographic conditions	2	3
8. Contaminants (pollution)	2	3
9. Contaminants in fish	1	2
10. Marine litter	2	4
11. Energy introduction	2	2
TOTAL	**29**	**56**

For Descriptor 1, the most Criterion and Indicator 'rich' of all the Descriptors, the complexity of the integration process is further compounded by the fact that the Indicator types proposed in the Decision document need to be applied to different ecosystem components. For the IA2017, for example, OSPAR proposes five separate ecosystem components: Pelagic Habitats, Benthic Habitats, Fish and Cephalopods, Birds and Reptiles, and Marine Mammals. However, the situation is slightly simpler than it might have been. It is not a straight forward cross-tabulation of five ecosystem components against 14 Indicator types, potentially giving 70 individual metrics, because the five ecosystem components consist of two habitat-type and three species-type components and the Decision documents distinguishes between species, habitat and ecosystem-level Criteria and Indicators. Table 7 maps proposed Indicator types to ecosystem components; this suggests that the maximum number of metrics might in fact only be 35. The table also suggests that, generally, only two criteria are relevant to each ecosystem component.

Table 7. Mapping of relevant indicator types listed in the EC Decision document for Descriptor 1 (EC, 2010) against the five ecosystem components for which subregional scale assessments of status will be required across the Northeast Atlantic MSFD region. PH = Pelagic Habitats; BH = Benthic Habitats; FC = Fish and Cephalopods; BR = Birds and Reptiles; MM = Marine mammals.

Level	Criterion	Indicator type	PH	BH	FC	BR	MM
Species	1.1 Distribution	1.1.1 Range			X	X	X
		1.1.2 Pattern within range			X	X	X
		1.1.3 Area covered by species (sessile/benthic species)		X			
	1.2 Pop. size	1.2.1 Abundance/biomass			X	X	X
	1.3 Pop. condition	1.3.1 Demographic characteristics			X	X	X
		1.3.2 Genetic structure			X	X	X
Habitat	1.4 Distribution	1.4.1 Range	X	X			
		1.4.2 Pattern within range	X	X			
	1.5 Extent	1.5.1 Area	X	X			
		1.5.2 Volume	X	X			
	1.6 Condition	1.6.1 Condition of typical species/communities	X	X			
		1.6.2 Relative abundance/biomass	X	X			
		1.6.3 Physical, hydrological and chemical conditions	X	X			
Ecosystem	1.7 Structure	1.7.1 Relative proportions of components (habitats/species)	X	X	X	X	X

These considerations suggest that a multi-level step-wise integration process will be required to fulfil IEA requirements under the MSFD. Other authors have reached similar conclusions, suggesting that to meet MSFD needs fully, at least four levels of integration will be needed (Cardoso *et al.*, 2010; Borja *et al.*, 2014). Our evaluation of integration needs under the MSFD suggests that four levels of integration will be required simply to arrive at an assessment of status for each individual Descriptor (Figure 13):

- Level 1 considers the assessment of the status of a single ecosystem component (e.g. fish communities) using a single indicator (e.g the large fish indicator (LFI)) across a single MSFD Subregion (e.g. the Greater North Sea), where this Subregion has been partitioned into a number of separate subdivisions reflecting spatial variation in the structure and composition of the ecosystem component in question. Analytical assessment of the indicator is undertaken at the subdivisional spatial scale. The outcomes of these individual subdivision assessments are then integrated to determine status at the scale of the subregion.
- Level 2 considers the assessment of the status of a single ecosystem component (e.g. fish communities) based on a single Criterion. For example, Criterion 1.1 “Distribution” may have been assessed using two indicators, ‘1.1.1 distribution range’ and ‘1.1.2 distribution pattern’ within the range. Since the both indicators in this particular example relate to the whole population of the species in question within the subregion, the analysis would most likely have been performed at the subregional scale; no level 1 integration of separate subdivisional scale assessments would

therefore be needed. However, the outcomes of the two separate distribution indicator assessments need integration to determine status at the Criterion level.

- Level 3 considers the assessment of the status of a single ecosystem component (e.g. fish communities) at the Descriptor level. Extending the Level 2 example above, the fish community in the subregion may also have been assessed using indicators '1.2.1 abundance/biomass' (for Criterion 1.2 "Population size"), '1.3.1 demographic characteristics' (for Criterion 1.3 "Population condition"), and '1.7.1 relative proportions of components' (for Criterion 1.7 "Ecosystem structure"). Indicators 1.2.1 and 1.3.1 both relate to whole populations within subregions so level 1 integration would probably not be necessary. But community-level metrics, such as the LFI, are the most likely candidates to fulfil the indicator 1.7.1 function. Their analysis will generally need to take account of spatial heterogeneity within subregions, and so will frequently be performed at a subdivisional spatial scale and require Level 1 integration. Since a single indicator has been assessed for each of these three additional Criteria, no level two integration is necessary to determine Criterion level status, but assessment outcomes for all three Criteria, along with that of Criteria 1.2 "Distribution" described above for Level 2, need integration to derive a subregional-scale Descriptor-level assessment for the fish community.
- Level 4 integration provides the overall assessment of status for each qualitative Descriptor of GES. This level of integration will only be needed for Descriptors where indicators for different ecosystem components have been monitored and assessed. In Describing integration levels 1 to 3, we have considered only a single ecosystem component in our examples, the fish community. But identical IEA procedures will also have been applied to the other ecosystem components: Pelagic Habitats, Benthic Habitats, Seabirds and Reptiles, and Marine Mammals. Level 4 integration combines the Descriptor-level assessments for each ecosystem component to derive an overall assessment for the Descriptor.

If a single overall evaluation of status is wanted, integrating the information conveyed by all eleven Descriptors, then a fifth level of integration will be necessary. In the framework described above we have assumed a logic of assessing each ecosystem component to Descriptor level and then integrating across ecosystem components. Within Descriptor 1 for example, the status of each individual ecosystem component (e.g. Benthic Habitats, Fish and Cephalopods, etc.) is first assessed based on the integrated outcomes of the assessments of individual indicators and Criteria, at subdivisional and subregional spatial scale (levels 1 to 3 of integration shown in Figure 13). Integration of the outcomes of the overall assessments for each ecosystem component then provides the ultimate assessment of status for Descriptor 1 (the fourth level of integration shown in Figure 13). The alternative would be to integrate across ecosystem components at either the Indicator or Criteria level. However, this would place the emphasis on these arbitrary human constructs rather than the natural biological entities, providing integrated assessments of status for each Criterion, for example, but not providing overall integrated assessments of status for each of the five ecosystem components.

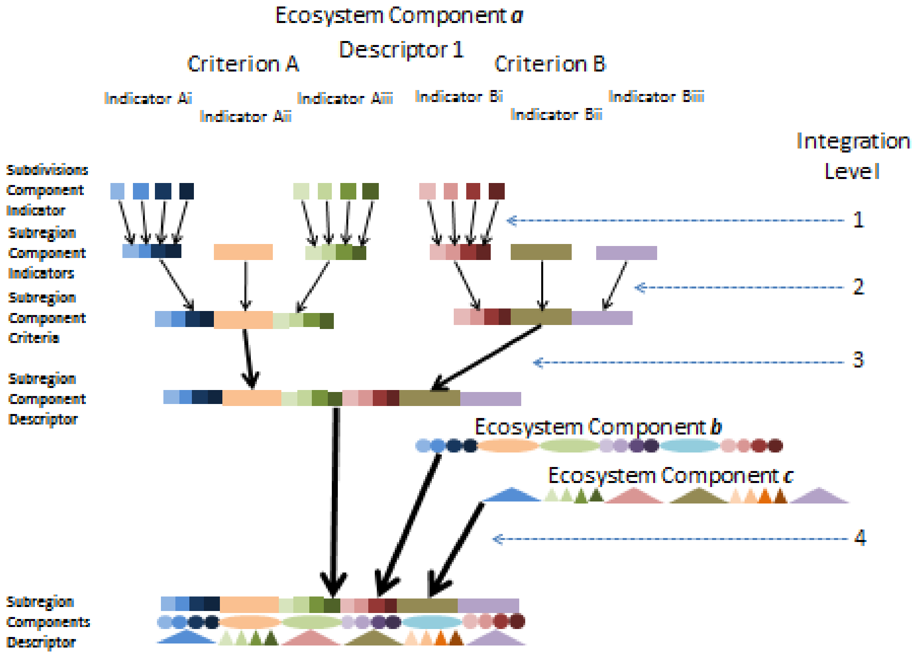


Figure 13. Schematic illustrating four levels of integration required to assess status at the level of single qualitative Descriptor of GES. The IEA needed to assess status for Descriptor 1 is illustrated, as this is likely to be the most complex.

Integration/aggregation methods

A critical question that has to be addressed in developing an IEA procedure to support MSFD implementation is how exactly the assessment outcome information should be combined at each integration phase. It seems probable that information aggregation methods will differ between indicators, between Descriptors, and between integration levels. Borja *et al.* (2014) consider nine possible methods for combining or aggregating the results of individual indicator assessments to derive an overall IEA outcome. Here we briefly review these different 'rules' to consider which of them might be used in an IEA to support EBM required under the MSFD.

The 'one out, all out' (OOAO) method

This is used in the Water Framework Directive (WFD) to integrate within and across Biological Quality Elements in order to determine the ecological quality status of each water body under consideration (CIS, 2003). Use of this integration method implicitly assumes all 'indicators/metrics' involved in the assessment to be equally weighted, and that overall status is equal to the 'indicator/metric' with the lowest status assessment score; any element with a sub-GES assessment score results in an overall assessment outcome of 'failing GES'. The approach therefore produces a 'worst case' outcome, has a high probability of producing a 'failing GES' outcome, and might therefore be considered to be precautionary. It has been suggested that the OOAO is most appropriate when the integration involves 'indicators/metrics' that monitor the effects of different pressures on the ecosystem (CIS, 2003; Caroni, 2013). Breen *et al.* (2012) used a worst-case OOAO approach to determine overall status for Descriptors, 3, 4, 8, 9, 10 and 11 (fish and shellfish, food webs, contaminants in the environment, contaminants in fish and shellfish, marine litter and underwater noise respectively). The OOAO method is particularly appropriate when legal criteria are involved, for example contaminants exceeding legally permitted levels, or habitats failing favourable conservation status under the Habitats Directive (Borja *et al.*, 2014). It is less useful where indicator precision is low (high uncertainty), so the risk of any one indicator/metric giving a false negative (see section on signal theory) assessment score, and so constituting the 'worst-case', is therefore high. The OOAO method may also be less appropriate in situations where some of the indicators used in the assessment might be less well developed (Borja *et al.*, 2014).

The 'scoring/rating' method

This approach assigns categorical scores (e.g. 1 = failing GES, 2 = failing GES, but improving situation, 3 = GES) to each 'indicator/metric' assessment outcome. The scores are summed and rated depending on the number of 'indicators/metrics' involved. The individual assessments can be weighted based on their considered importance to ecosystem functioning and services (e.g. based on the outcome of a formal risk assessment). This approach provides the basis for many multi-metric indices used in the WFD (Borja *et al.*, 2004; Birk *et al.*, 2012). It has also been trialled in a cross-descriptor integration combining the eleven MSFD Descriptors. An ecological quality ratio (EQR) was determined for each indicator used for each Descriptor. The EQR for each Descriptor was then determined as the average of the relevant indicator EQRs. The Descriptor EQRs were then multiplied by a weighting factor (weighting the importance of each Descriptor in determining overall environmental status – an expert opinion judgement) and summed to derive an overall environmental status value (Borja *et al.*, 2010; Borja *et al.*, 2011b). Problems might arise when using this integration method if the indicators involved are sensitive to different pressures (Borja *et al.*, 2014). This method may be more useful in situations where some of the 'indicator/metrics' used in the assessment are less well developed than others; these can be given lower weighting (Borja *et al.*, 2014)

The 'multi-metric' method

The method integrates multiple metric values to derive a single indicator value, which can then be compared against a single target value determine *a priori*. The WFD relies heavily on multi-metric indices to assess overall water-body quality (Birk *et al.*, 2012). The JRC/ICES Task Group 6, tasked with proposing indicators to support implementation of the MSFD in respect of "seafloor integrity", recommended the use of multi-metric indices or multivariate techniques for integrating indicators of benthic invertebrate species composition, diversity and distribution, etc. (Rice *et al.*, 2010). This approach is considered by some to provide more robust results compared with indicators based on single parameters (Borja *et al.*, 2011a; Borja *et al.*, 2014). Often trends in individual single-parameter indicators are correlated (Greenstreet *et al.*, 2012b), and this is not a good trait in indicators being combined into a single multi-metric index (Borja *et al.*, 2014). Scaling of each multi-metric index could also present some problems (Primpas and Karydis, 2011). Again, difficulties may arise if the individual metrics contributing to the multi-metric indicators are sensitive to different pressures (Borja *et al.*, 2014). The 'multi-metric' method is particularly suitable if integrating several similar indicators, for example species composition, or several indicators of eutrophication (Borja *et al.*, 2014).

The 'averaging' method

Borja *et al.* (2014) suggest that this is the most commonly used approach, using simple calculations such as arithmetic average, weighted average, hierarchical average, median, sum, product, or combinations of these method, to derive a single overall assessment outcome using the values observed for a number of different 'indicators/metrics' (Shin *et al.*, 2012). Most of these methods again assume that the 'indicators/metrics' involved are equally weighted; the exceptions being weighted and hierarchical averaging, but these methods depend on having a robust, defensible basis for setting the weighting or hierarchy. Where 'expert opinion' is used to determine the weighting or hierarchy, this may not always be consistent (Aubry and Elliot, 2006), and in circumstances where the basis for deciding the weighting or hierarchy is not definitive, use of one of the simpler arithmetic averaging methods is recommended (Ojaveer and Eero, 2011). The averaging method is particularly applicable where the indicators involved are sensitive to the same pressure (Borja *et al.*, 2014).

The 'multi-dimensional' method

This method uses multivariate methods, such as discriminant analysis or factor analysis, to combine parameters within a multi-dimensional space. Prior to carrying out the IEA, this multi-dimensional space will need to be partitioned into areas representing GES and areas representing failing GES, but specific targets may not be required for the individual indicators being integrated in this way (Borja *et al.*, 2014). However, interpretation is less intuitive and the results more difficult to communicate to managers and stakeholders. The information provided by the individual indicators involved in the assessment is subsumed in the process and therefore lost to some extent, and this may make the provision of specific management advice more difficult (Shin *et al.*, 2012). The 'multi-dimensional' method is particularly suitable if integrating several similar indicators, for example species composition, or several indicators of eutrophication (Borja *et al.*, 2014).

The 'conditional rules' method

This method can take a variety of forms, but all imply that the 'conditional rules' be specified *a priori* of carrying out the IEA. Conditional rules are set whereby specific criteria are met, and expert judgement can be used to set these criteria. Examples include the 'conditional rule' for benthic community condition under Descriptor 6 (seafloor integrity) in Hellenic waters that at least two out of three indicators (one biotic and two structural or diversity indicators) should meet their individual targets (Simboura *et al.*, 2012). An extension of this rule could be to suggest that the single biotic index must meet its target, and that only one or other of the two structural/diversity indicators might be allowed to miss its target. Tueros *et al.* (2009) provide a similar example of the use of the 'conditional rule' method when integrating water condition and sediment condition indicators to derive an overall assessment of the chemical status of water bodies under the WFD. Breen *et al.* (2012) use risk criteria 'conditional rules' to integrate the results of individual indicator assessments to derive overall assessments of status for Descriptors 1, 2, 5 and 6: biodiversity, non-indigenous species, eutrophication and seafloor integrity respectively. The use of 'conditional rules' has the advantage of focusing attention on key elements contributing most to ecosystem health, but it assumes that GES can be determined on the basis of the few 'indicators/metrics' related to these key elements.

The 'decision tree' method

This method is closely related to the 'conditional rules' approach, but provides the opportunity of applying different rules when combining individual 'indicator/metric' assessments to derive an overall assessment. Arbitrary decisions can be applied at each step within the decision tree; these decision rules can be qualitative, quantitative, or based on expert judgement, thereby providing considerable flexibility in reaching a final overall assessment. However, the decision rules applied at each step in the process should be specified prior to commencing the assessment exercise. This method can allow for greater incorporation of ecosystem complexities, interactions and feedback loops. Borja *et al.* (2013) implicitly use a 'decision tree' approach when deciding that in order for marine biodiversity (Descriptor 1) to be at GES, all other Descriptors must be at GES and if a pressure-related Descriptors fails GES, then D1 also fails. Borja *et al.* (2004; 2009b) also use a 'decision tree' type approach to integrate biological indicators covering a range of taxa with physicochemical indicators to perform an assessment overall status; they consider the approach used to be compliant with both MSFD and WFD requirements.

The 'probabilistic' method

Actual status of any ecosystem element is rarely, if ever, known with absolute certainty; individual 'indicator/metric' values calculated from the monitoring data represent the actual status of the ecosystem elements monitored, but with a degree of uncertainty, the extent of which is often unknown. If the degree of uncertainty can be approximated, then this information can be used when integrating the different 'indicators/metrics'. Greater weight can be assigned to 'indicators/metrics' deemed to be the least uncertain. Bayesian statistics can be applied to derive the overall assessment score, providing coherent and transparent rules by which the final assessment outcome is arrived at (Borgia *et al.*, 2014). The 'probabilistic method' can be combined with several of the methods previously listed, for example linked with the use of 'conditional rules' or 'decisions trees'. This approach has been applied within the DPSIR framework to manage eutrophication (Barton *et al.*, 2008; Barton *et al.*, 2012) and to assess the severity of oil spills (Lehikoinen *et al.*, 2013b). A major advantage of the 'probabilistic' method lies in the fact that a probability could also be assigned to the final overall assessment outcome; for example, GES has been attained with a probability of 70%. The question then would be how much uncertainty managers would be able to tolerate (Borja *et al.*, 2014). A disadvantage of this method is that it is data demanding, complex to calculate, and would almost certainly be quite difficult to communicate to managers and stakeholders.

The 'high-level integration' method

This centres on the use of 'assessment tools' such as the HELCOM Holistic Assessment of Ecosystem Health Status (HOLAS) tool. The tools combine some of the methods listed above, such the 'conditional rules' and 'OOAO' methods, to integrate between different sets of indicators to arrive at an overall status assessment. It generally uses a reduced set of what are considered to be key indicators within each indicator suite, and applies weightings to these indicators that have been agreed previously. One example includes an integrative method using a weighted scoring or rating approach applied in the southern Bay of Biscay to meet MSFD needs (Borja *et al.*, 2010; Borja *et al.*, 2011b). Halpern *et al.* (2012) propose an alternative method, based primarily on human pressures and activities, which illustrates high-level integration at national level. Micheli *et al.* (2013) have used a similar method to assess the impact of cumulative human activities on Mediterranean and Black Sea ecosystems. Tett *et al.* (2013) have also developed a 'high-level integration' procedure for the North Sea that involves five separate calculation steps:

i. Define spatial extent of assessment

ii. Determine spatial granularity and temporal frequency of sampling

iii. Select state variables

iv. Plot indicator trajectories in state space and calculate Euclidean distance from (arbitrary) reference condition

v. Calculate medium-term variability about trend in state space and use inverse as a proxy for resilience

Step iv aligns closely with the 'multi-dimensional' integration method listed above. At the final level of integration when performing a IEA to support MSFD, i.e. when integrating across the 11 Descriptors, the principal anthropogenic pressures on the ecosystem could be taken into consideration; if subject to heavy fishing pressure, Descriptors 1, 2, 3, 4, 6, and maybe 11 are likely to be most affected and these Descriptors could be assigned higher weighting than Descriptors 5, 7, 8, 9, and 10 (Borja, *et al.*, 2014). Methods of integrating across the state Descriptors will in all likelihood vary from those used to integrate across the pressure Descriptors (Borja *et al.*, 2014).

Considerations for selecting integration/aggregation methods at each integration level

Level One

Level one considers the assessment of the status of a single ecosystem component using a single indicator across a single MSFD Subregion/Region, where this Subregion/Region has been partitioned into a number of separate sub-divisions reflecting spatial variation in the structure and composition of the ecosystem component in question. An example of such an assessment process using the LFI to assess the status of the fish community in five subdivisions of the Greater North Sea Subregion has been the subject of a recent study (Greenstreet *et al.*, in press), who argued that a one out-all out (OOAO) approach should be adopted; the indicator used should meet individual targets set for each spatial sub-division in all subdivisions. They suggest that it would be illogical to maintain that the fish community was at GES at regional-scale, i.e. across the whole Greater North Sea, in a situation where, should the indicator target fail to be achieved in one subdivision, this was demonstrably not the case in approximately 20% of the Subregion area.

Other views are, however, equally valid. There are potential pitfalls in rigidly adhering to an OOAO rule at level one: the risk of false negatives for example. Confidence limits are not currently set on the LFI. If in one subdivision the LFI missed its target by only 5%, then under a OOAO rule this would result in an overall 'failing' assessment at the Subregional/regional scale. However, precision in estimating the LFI could easily be ± 10%. Without confidence limits, we have no knowledge of the actual level of precision and missing a target by 5% could in reality not represent a statistically significant 'failure'. Application of a 'conditional rule' could address such a situation. If the target for the LFI is to exceed a specified value then, assuming that error around the estimate is symmetrical, when the observed LFI exactly equals its target there would be a 50% probability of the actual LFI exceeding the target and a 50% probability of it being below the target. The binomial distribution therefore suggests that if the LFI exceeds its target five years (or more) out of every six, then this could be construed as statistically significant evidence that the target in any given subdivision had been achieved. Conversely, failing the target in five years (or more) out of six should be construed as statistically significant evidence of a missed subdivision target and the OOAO rule should then apply in deriving the integrated assessment for the Subregion/Region. Failing any subdivision target in one to four years (so meeting the target in one or two years) in any six year period should not necessarily cause an overall 'failing' assessment at the Subregional/regional scale, but here the 'conditional rule' should require the number of such failures to decline in subsequent six-year assessment periods, so that the trend was towards a statistically significant 'meeting target' assessment.

Alternatively, if an indicator target was just missed in one subdivision, but comfortably met in the remaining subdivisions, then this could be construed as justifying an overall 'GES' assessment at the Regional/Subregional scale. The LFI, for example, is a quantitative indicator, so some form of 'averaging' method could produce such an outcome if deserved. However, straightforward averaging of LFI values across the individual subdivisions (s) that make up the whole subregion (S) is unlikely to be appropriate because targets would almost certainly not be the same in each subdivision. Spatial heterogeneity in the fish community would probably be the reason for carrying out the assessment analysis at a subdivisional scale in the first place, so separate targets appropriate to the fish assemblage resident in each subdivision would need to be established. Observed LFIs in each subdivision ($LFI_{s,obs}$) would therefore first have to be scaled by their relevant target ($LFI_{s,targ}$) prior to averaging across subdivisions. The resulting 'average LFI' (LFI_{avg}), calculated as:

$$LFI_{avg} = \frac{\sum_{s=1}^{s=S} \frac{LFI_{s,obs}}{LFI_{s,targ}}}{S}$$

would now scale between 0 and rarely likely to exceed 2 to 3 depending on individual targets, but in all cases, a value of $LFI_{avg} > 1.0$ would represent an overall assessment of 'GES' at the Subregional/Regional scale.

Level Two

Level two considers the assessment of the status of a single ecosystem component within a single MSFD Subregion/Region at the Criterion level. If only a single indicator has been used to monitor change in the Criterion, then integration is not required; the Criterion level assessment outcome is the same as the outcome for the individual indicator assessment. If however, two or more indicators have been monitored and assessed for any given Criterion, then the outcomes of these individual indicator assessments will require integration to derive the overall Criterion level assessment. The integration/aggregation method used will depend on the indicators. A key consideration might be whether or not the indicators concerned are sensitive to the same pressures.

For example, if assessing the state of the fish community against Criterion 1.1 "Distribution", both indicators, '1.1.1 distribution range' and '1.1.2 distribution pattern' within the range, might be used. Both are likely to be similarly affected by variation in fishing pressure; range might decline and the population might become more restricted to just part of the range giving rise to likely reductions in the value of both metrics. Several options for integration could be considered. Firstly a OOAO rule could be applied, effectively attributing equal weighting to both aspects of a species distribution and, in a worst case scenario, requiring both aspects to meet their targets to attain a 'GES' assessment outcome. Since both distribution aspects tend to be correlated, so long as the two targets have been consistently set, then if one indicator meets its target, the second should soon do so as well. This same logic might also suggest the use of a condition rule requiring only one of these two indicators need meet its target, perhaps so long as the second indicator is at least showing the necessary trend towards attaining its target value. Alternatively, an averaging rule could be applied, which would permit one indicator to miss its target so long as the second indicator achieved its target with sufficient excess as to compensate for the first indicator's failure. A weighting procedure could also be applied to the averaging process, inferring greater importance to the indicator deemed to be most critical. These last three options provide some leeway in the integration process to allow for a degree of inconsistency in target setting.

If using an averaging method, then almost certainly some form of scaling of individual indicator values would be necessary, again presumably using each indicator's target value as the scaling parameter as in the example above for the LFI. This would again produce a Criterion level assessment outcome metric that would range between 0 and some positive value >1.0, where any metric value >1.0 would suggest a 'GES' assessment outcome.

In setting targets for commercial fish stock state indicators within an MSFD context, one would need to take account of the fact that you could only realistically expect to be at the maximum sustainable yield biomass level (B_{msy}) in 50% of years if fishing at the maximum sustainable fishing mortality rate (F_{msy}), but this can be taken into account in the wording of the state indicator target: i.e. stock biomass should equal or exceed B_{msy} in three years in each six-year MSFD assessment cycle. Conversely, the target could remain that stock biomass should equal or exceed, and a conditional rule applied during the integration process, such that failure to achieve an overall "GES" outcome would only occur if this particular indicator missed its target in more than three years in each six-year assessment period.

Level Three

Level three considers assessment of the status of a single ecosystem component at the Descriptor level, integrating the Criterion-level assessment outcomes. Thus, for example, assessing fish communities against Descriptor 1 could require integration of up to four separate Criterion level assessment outcomes: for Criteria '1.1 Population Distribution', '1.2 Population Size', '1.3 Population Condition', and '1.7 Community Structure'.

In this particular example, variation in the assessment outcomes for the two Criteria, '1.2 Population Size' and '1.1 Population Distribution' are likely to co-vary because of the well-established ecological relationship between abundance and distribution (Rindorf and Lewy, 2012); the more abundant a species is, the more widely distributed it will be

(Gaston *et al.*, 2000; Blackburn *et al.*, 2006). This pattern holds both within species (as a species increases in abundance, so its range increases) and between species (more abundant species occupy larger ranges than rare species), and both intra- and inter-specific abundance-occupancy relationships have been widely demonstrated in fish populations (Overholtz, 2002; Hinz *et al.*, 2003; Fisher and Frank, 2004; Blanchard *et al.*, 2005). In such circumstances a number of integration options are logically available:

- a conditional rule might be applied, requiring one or other Criterion attain a 'GES' assessment outcome;
- or an averaging integration method might be used, perhaps assigning greater weighting to the '1.2 Population Size' Criterion level assessment outcome, since population size is the principal driver in the abundance – distribution relationship;
- or a OOAO rule might also be applied, given the likelihood that if target setting has been consistent, if one Criterion achieves a 'GES' assessment outcome, the second should soon follow.

Application of the OOAO rule at this level of integration confers equal weighting to each Criterion. This could be justifiable, or conversely sound reasons might exist to suggest that greater weight should be given to some Criteria over others. The MSFD Decision document lists 7 Criteria for Descriptor 1; at some stage presumably, the authors thought all seven Criteria to be important. Scientists on the other hand might hold the view that, with respect to maintaining or restoring biological diversity, some Criteria are more critical than others. Thus for example, Criterion '1.2 Population Size' is currently monitored using the 'abundance of a suite of sensitive species indicator' (Greenstreet *et al.*, 2012). It could reasonably be argued that ensuring recovery in the populations of these threatened and vulnerable species contributes far more to the restoration of biological diversity among fish, than recovering fish community size structure, the attribute of the fish component monitored by the LFI, which is currently used to assess status for Criterion '1.7 Ecosystem/Community Structure'. Such a view might be further reinforced if taking account of the time-lags involved in the relation between variation in the LFI and fishing mortality (Greenstreet *et al.*, 2011; Shepherd *et al.* 2011; Modica *et al.*, 2014). Such a situation might again represent a prime example where a well-considered condition rule might be most useful.

At several points so far we have considered the need to standardise individual indicator values against their target values to derive assessment metrics that range between 0 and a relatively small number greater than 1.0, and where any resulting assessment metric with a value >1.0 therefore represents a 'meeting GES' assessment outcome. Such a process generates assessment metrics that permit integration or aggregation of indicators that vary across very different scales: an LFI of 0.21 relative to a target of 0.3 gives an assessment metric of 0.70 and this can be directly compared, averaged, aggregated or integrated, with an assessment metric of 1.15 derived from a count of 15 sensitive species showing recovery in their population size against a target of 13 species required to do so. In this instance, a straightforward averaging integration procedure generates an integrated assessment outcome across the two indicators of 0.93: a 'failing GES' outcome. Given the arguments stated above, however, a weighting factor of 1.3 might be assigned to the 'abundance of a suite of sensitive species' indicator (giving this indicator 30% more importance in the IEA) and this would result in an integrated assessment outcome of 1.10: a 'meeting GES' outcome. In this example we have considered two indicators, but in this instance these are the only indicators used to monitor status in each Criterion. Thus no Level two integration would be required and this example is appropriate to Level three, because it consequently addresses the integration to the Descriptor level: aggregating the two Criterion Level assessments. One key message emerges from this example; the resulting integrated assessment outcome metrics, be it 0.93 or 1.10, need no further standardisation, they are already in a form that can support further aggregation to higher integration levels.

Difficulty arises, however, where assessment or integration gives rise to non-numeric categorical outcomes, e.g. categories such as 'failing GES' (score A), 'failing GES, but recovering' (score B) and 'meeting GES' (score C), as this places restrictions on further integration to higher levels. Few integration methods are available to integrate such outcomes and if one such categorical outcome is involved in an integration process that includes numeric outcomes. The higher the level of integration, the more likely it is that categorical outcome could become involved. Arithmetic integration methods become difficult, if not impossible if categorical assessment outcome are involved. Any numeric assessment outcome might first have to be converted to a categorical outcome to allow the scoring/rating method to be applied, or else the OOAO approach or a conditional rule might have to be adopted.

Level Four

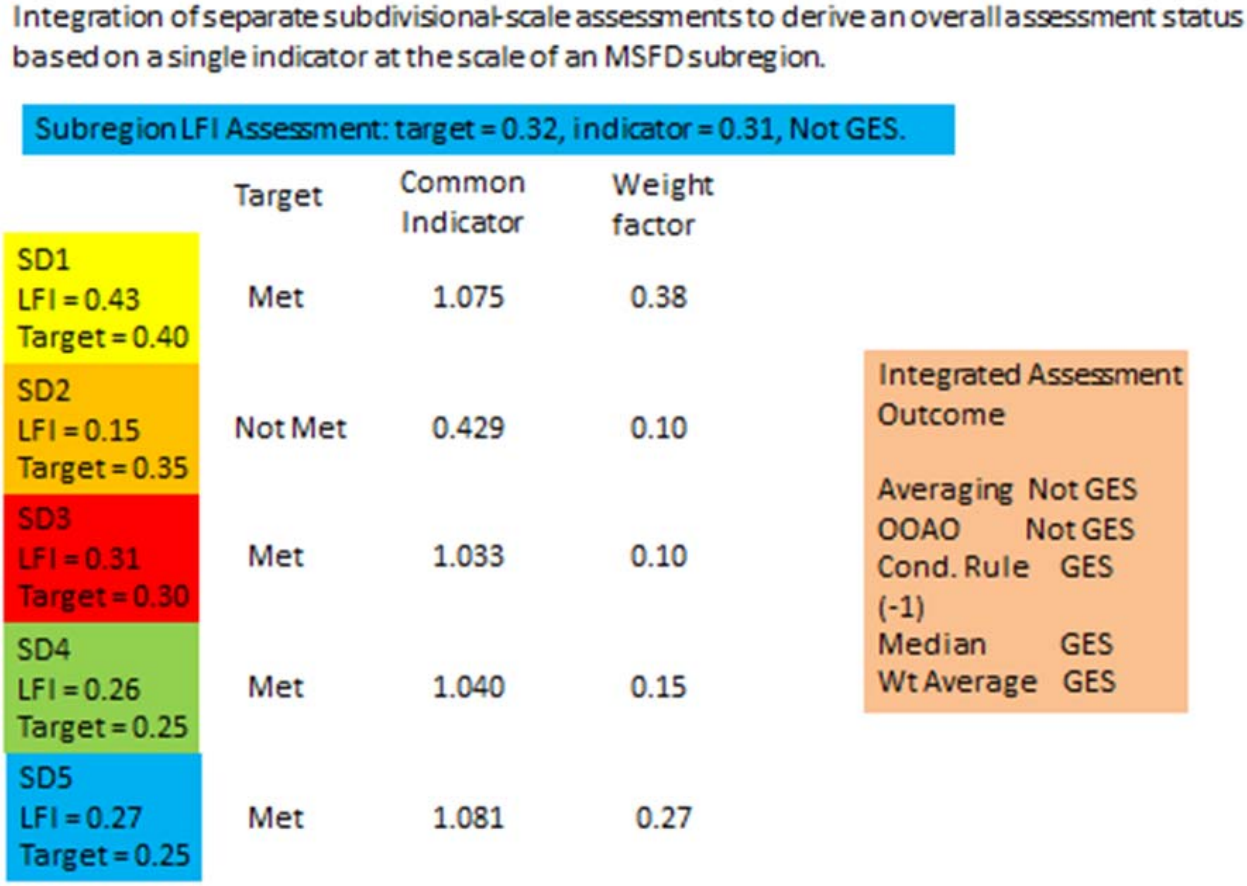
Level four integrates the Descriptor level assessment outcomes for each ecosystem component to derive an overall assessment of status for the Descriptor. At this level, each ecosystem component will have been subjected to assessment through Levels one to three. Each ecosystem component will therefore independently be deemed either to be at 'GES' or 'failing GES' at the Descriptor level. At level four it seems inconceivable that anything other than a OOAO rule should apply; if at the scale of a MSFD Subregion, the entire Greater North Sea for example, one ecosystem component, seabirds for example, is deemed to be below GES at the Descriptor level, it would seem counter to the spirit, if not the letter, of the MSFD to consider that the Subregion in question could be anything other than 'failing GES'.

How does choice of integration method affect overall IEA outcome?

Several authors have commented on the importance of using the most appropriate integration methods because this can have a profound influence on the eventual IEA overall outcome (Ojaveer and Eero, 2011; Borja *et al.*, 2013; Caroni *et al.*, 2013). In this section we present the results of two simulations. The first explores the consequences of different integration methods applied to a specific hypothetical situation in a Level One integration. The second is a more generic simulation of the effect of different integration approaches across several integration levels.

Consequences of integration method choice at a Level one integration based on the LFI

Figure 14 illustrates a simulated Level one assessment based hypothetically on the large fish indicator used to assess the status of the structure of the fish community in terms of its size composition in the Greater North Sea. Taking account spatial heterogeneity in the species composition of the demersal fish community, the subregion in question is split into five subdivisions (see Figure 7) and separate analytical evaluations are made for each subdivision based on LFIs observed in each subdivision compared against individual subdivisional targets. A single overall subregional scale assessment is also applied for comparison. Figure 14 indicates the different subregional scale assessment outcomes when five different aggregation/integration methods are applied. The condition rule simply allows for one of the five subdivision assessment to give a 'Not GES' outcome. Weightings given to each subdivisional assessment could be based on subdivision area. In this case hypothetical weightings were applied that weighted Subdivisions 1 through to 5 38%, 10%, 10%, 15% and 27% of the total respectively. The subregional scale assessment outcome is strongly dependent upon the aggregation/integration methodology applied. A single subregional scale indicator assessment also gives a "Not GES" outcome; the same as achieved using the OOAO rule or an "averaging" integration methods.



Integration of separate subdivisional-scale assessments to derive an overall assessment status based on a single indicator at the scale of an MSFD subregion.

Subregion LFI Assessment: target = 0.32, indicator = 0.31, Not GES.

	Target	Common Indicator	Weight factor
SD1 LFI = 0.43 Target = 0.40	Met	1.075	0.38
SD2 LFI = 0.15 Target = 0.35	Not Met	0.429	0.10
SD3 LFI = 0.31 Target = 0.30	Met	1.033	0.10
SD4 LFI = 0.26 Target = 0.25	Met	1.040	0.15
SD5 LFI = 0.27 Target = 0.25	Met	1.081	0.27

Integrated Assessment Outcome

Averaging	Not GES
OOAO	Not GES
Cond. Rule (-1)	GES
Median	GES
Wt Average	GES

Figure 14. Integration of separate subdivisional-scale assessments to derive an overall assessment status based on a single indicator at the scale of an MSFD subregion and showing the effect of using five different aggregation methods. Deriving the "common indicator" is the first step in the aggregation process and this is simply done by dividing the observed indicator value by its target value.

A simulation to integrate GES from indicator level to overall status at the Descriptor level

Here the effects of different aggregation methods at Indicator level, Criterion level and Descriptor level and across Descriptors are explored to assess the consequences of using different integration methods to determine an overall IEA outcome. Due to the partly nested nature of several integration methods described by Borja *et al.* (2014), only four different methods were simulated. Note that this simulation does not fully match the hierarchical integration levels discussed so far. The simulation is presented for illustrative purposes only and some important general conclusions emerge.

Introduction

The MSFD is structured in to 56 Indicators clustered into 29 Criteria, which in turn are grouped into 11 Descriptors (EU-COM, 2010). Indicators, Criteria and Descriptors therefore reflect different hierarchical levels, and for each level the status has to be assessed with that aim of achieving GES at this level (Figure 15). The ultimate level of integration required remains uncertain, whether to individual Descriptor level or across all eleven Descriptors (Borja *et al.*, 2013), but for this simulation we have assumed the more arduous case, the latter, since there is some suggestion that this will be necessary (Claussen *et al.*, 2011; Cardoso *et al.*, 2010). This is the final integration level shown in Figure 15, which therefore shows the integration structure assumed in the simulation. Note that this excludes two integration levels illustrated in Figure 13, which are likely to be necessary. These are the integration that may be needed if analytical analysis of indicators is performed at the subdivisional spatial scale level, and the integration required across different ecosystem components at Descriptor level that will certainly be required for some Descriptors (e.g. Descriptor 1 Biodiversity and Descriptor 4 Food webs). Table 8 explains the abbreviations used in the simulation. We use the terms aggregation and integration synonymously, contrary to other authors (e.g. Borja *et al.*, 2014).

Table 8. Overview of abbreviations used.

Abbreviation	Meaning
$P(GES_I)$	Probability of an Indicator to achieve its assessment benchmark and thereby GES. Set to 0.4 by default.
$P(GES_C)$	Probability of a Criterion achieving GES (GES_C).
$P(GES_D)$	Probability of a Descriptor achieving GES (GES_D).
$P(GES_{TOT})$	Probability of achieving overall GES across all Descriptors (GES_{TOT}).

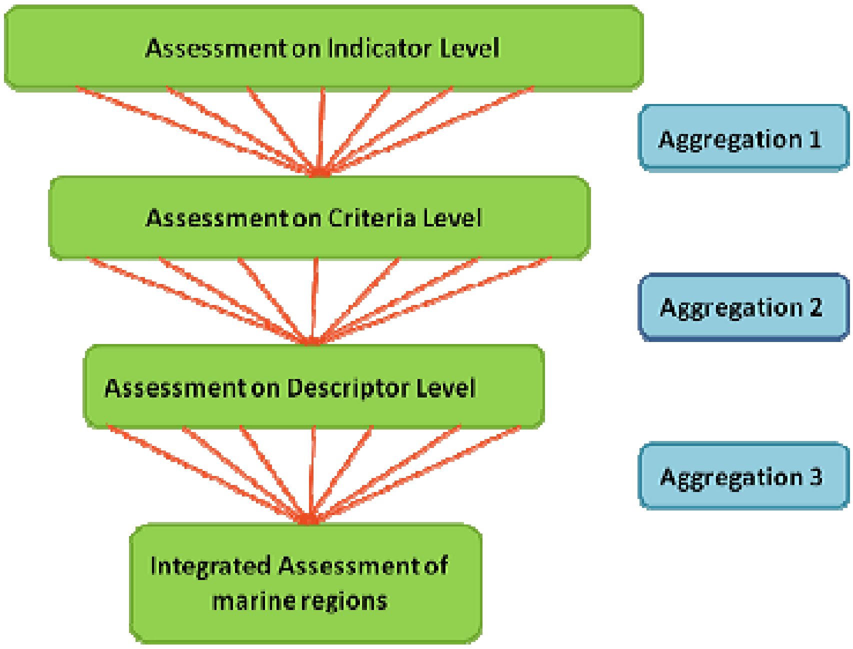


Figure 15. Possible aggregation levels for the integrated assessment of marine regions within the EU-Marine Strategy Framework Directive. From Claussen *et al.* (2011).

The reference data set

To compare the effect of different aggregation methods a reference data set containing 1000 random Monte Carlo runs was created. Each data set contained one value for each of the 56 MSFD Indicators modelled by a random uniform distribution ranging from 0 to 1. This distribution was chosen to simulate an absolutely random distribution of Ecological Quality Ratios (EQR), which also range from 0 to 1 (Borja *et al.*, 2011; Borja *et al.*, 2004). EQR represent the ratio between the environmental target (or a reference condition within the context of the WFD) and the current state. Thus EQR can be considered as a way to standardized non-uniform indicator metrics. The threshold of EQR between non-GES and GES was set at 0.6 (Borja *et al.*, 2011). Accordingly, an Indicator, Criterion or Descriptor was considered to achieve have achieved GES, if its score was > 0.6.

The aggregation methods

The status of single Indicators has to be integrated within Criteria, these in turn have to be integrated within Descriptors and finally, Descriptor level assessment outcomes may require integration to determine the final outcome of the overall IEA (Figure 15). The methods by which these different levels of integration are achieved can profoundly affect he assessment outcome at each level (Ojaveer and Eero, 2011). Here we simulate four different integration methods applied to a reference data set: the 'one-out-all-out'-rule (OOAO), arithmetic mean integration (AA), median integration (ME) and the application of a probabilistic approach (PROB) (Borja *et al.*, 2014).

If using the OOAO, GES is only achieved when all elements being integrated meet their GES targets. Thus all Indicators within a Criterion would need to achieve GES_I (with a score > 0.6 in our simulation) in order to achieve GES_C. At the Descriptor level, each Criterion would need to have achieved GES_C in order for the Descriptor to achieve GES_D and for GES_{TOT}, each Descriptor would need to be at GES_D in order to achieve overall GES (GES_{TOT}). The AA method integrates the single scores by taking the arithmetic means of Indicator, Criterion or Descriptor values (ranging from 0 to 1) within each Criterion or Descriptor and across all Descriptors, respectively. At each level of integration, the average score of Indicators, Criteria or Descriptors had to be > 0.6 in order to achieve GES at the next higher level of integration. The ME method was similar to the AA method but used the median instead of the arithmetic mean as integration mechanism. The probabilistic method was based on the binomial distribution which determines the probability that a number of successful trials (K) within a number of performed trials (N) occurs. To maintain the GES threshold of 0.6 at each hierarchical level, the minimum number of successful trials (K_{GES}) was chosen which was closest to the 0.4 probability (Table 9). This meant that if a Criterion contained three Indicators (N=3), at least two Indicators (K_{GES}=2) must meet GES_I to give a positive outcome for GES_C. Similarly, if a Descriptor contains seven Criteria (N=7), at least four Criteria (K_{GES} =4) had achieve integrated GES assessment outcomes to give a Descriptor level outcome of GES_D. Due to the discrete integer nature of the binomial distribution, the GES target of 0.4 is not possible for all N, so depending on N, the target probabilities ranged from 0.290 to 0.640 (Table 9).

Table 9.Number of total elements (N) and minimum number of elements required to meet GES (K_{GES}) necessary to give a probability of P(K>=PK_{GES}) that as closely as possible = 0.4. Here element can be Indicators within a Criterion, Criteria within a Descriptor, or the eleven MSFD Descriptors.

N	K_{GES}	P(K≥K_{GES})
1	1	0.400
2	1	0.640
3	2	0.352
4	2	0.525
5	3	0.317
6	3	0.456
7	4	0.290
8	4	0.406
9	4	0.517
10	5	0.367
11	5	0.467

The problem emerging from the integration process

At each integration level, i.e. when aggregating individual outcomes for Indicators, Criteria or Descriptors, the use of different integration methods resulted in different probabilities of achieving GES_C and GES_D (Figure 16). Using the OOAO, the probability of achieving GES within a Criterion or Descriptor reduces exponentially as the number of Indicators increases (Figure 16: A and B). The maximum number of Indicators listed for any given MSFD Criterion is four (Criteria C3.3, C5.2 and C6.2). For such Criteria, the probability of achieving GES at the Criterion level ($P(GES_C)$) is lower than 0.03 ($=0.4^4$) if using the OOAO. Similar exponential decreases of $P(GES_D)$ were evident as the number of Indicators and Criteria in integrations up to Descriptor level increased (Figure 16: C). Using the AA and ME integration methods, P(GES$_C$) and P(GES$_D$) still decreased as the number of Indicators or Criteria being integrated increased, but using these methods, the decrease was linear in form, although still at a marked rate of decline (Figure 16: D to I). Using the probabilistic integration method, no such trend was evident, but the probability of achieving GES also varied between the Criteria and Descriptor levels (Figure 16: J to L). This was due to the discrete integer nature of the binomial distribution which meant that no K$_{GES}$ could be determined such that P(K≥K$_{GES}$) was exactly =0.4. Instead K$_{GES}$ was selected as the minimum number of elements closest to 0.4.

The probability of achieving GES at the Descriptor level varied between Descriptors and between the four different integration methods (Figure 17). If using the OOAO to integrate, the probability of P(GES$_D$) was <0.2 for all Descriptors. P(GES$_D$) also failed the 0.4 benchmark for both the AA and ME methods, whereas the PROB method achieved this benchmark for all Descriptors except Descriptor 4 ‘Food webs’. None of the four methods therefore provided a consistent approach in which P(GES) remained constant between the hierarchical levels.

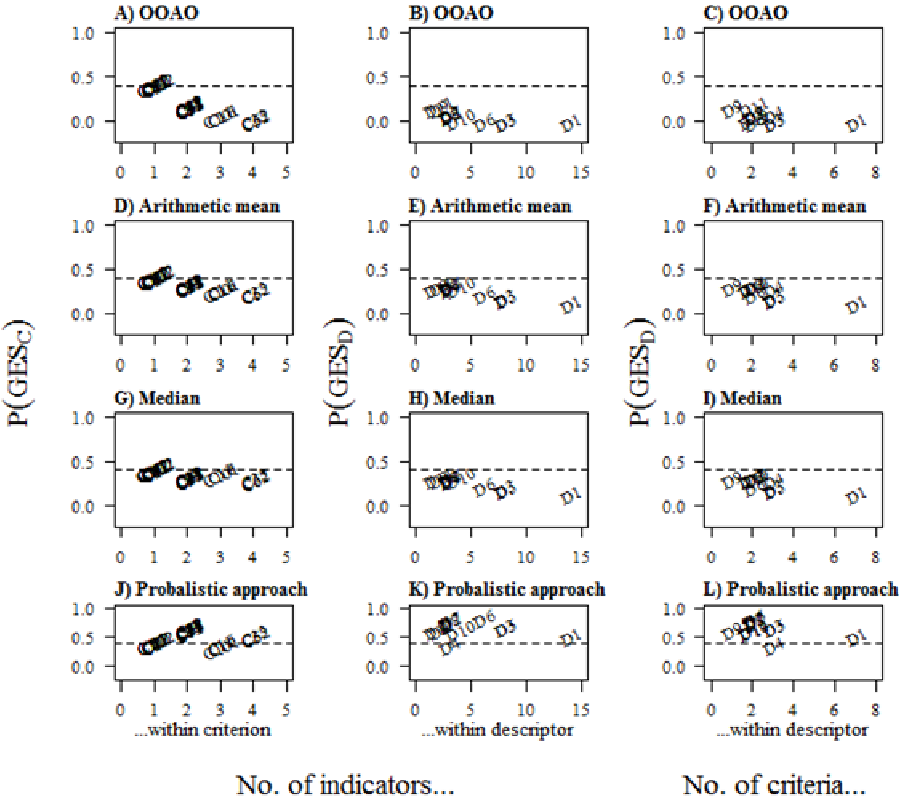


Figure16. Probability of achieving GES within MSFD criteria (P(GES$_C$)) or descriptors (P(GES$_D$)) in relation to the number of Indicators within a Criterion (A,D,G,J), Indicators within a Descriptor (B,E,H,K) or Criteria within a Descriptor (C,F,I,L) for four different integration methods.

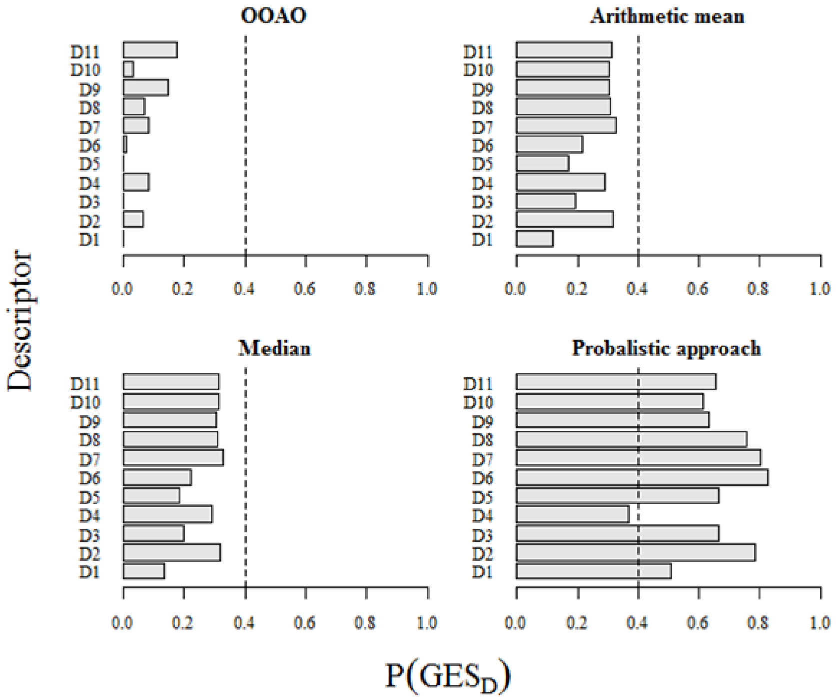


Figure 17. Probability of achieving GES$_D$ (P(GES$_D$)) with four different integration methods across the eleven MSFD Descriptors. The dashed line indicates the 0.4-probability benchmark which would indicate a consistent probability of achieving GES from Indicator to Descriptor level (P(GES$_I$) = P(GES$_I$) = P(GES$_I$) =0.4).

So why did the probabilities of achieving GES vary so much as the hierarchical level of integration increased? Using the OOAO approach, the probabilities of achieving GES multiplied, e.g. the probability of two Indicators achieving GES was $0.4^2=0.16$, of three Indicators $0.4^3=0.064$, so when integrating across multiple Indicators and Criteria, the probability of achieving GES approaches zero at the Descriptor level. Using the ME and AA approaches, variation in P(GES) at each integration level was related to predictions derived from the central limit theorem: means of means tend towards a normal distribution, thus the 0.4 quantiles alter as the distribution shifts across increasing hierarchical levels of integration (Figure 18). At the Indicator level, the distribution is random uniform (Figure 18: A), but approaches a normal distribution as the level of integration gets higher (Figure 18: B to D). While the mean of the distributions remains constant at 0.5, the probability of achieving a value >0.6 decreases from 0.4 to 0.019. Using the probabilistic approach resulted in a higher likelihood of achieving GES at higher integration levels. For most Descriptors this was partially explained by the inconsistent probabilities of K_{GES} (Table 9). The probability of achieving GES was closest to the 0.4 benchmark for Descriptor 4 because this Descriptor has only three Criteria with one Indicator each, therefore P(GES$_C$) = 0.4 and P(GES$_D$) = $(0.4^2*0.6)^3+0.4^3$ =0.352, which was close to the observed D4 mean score from the 1000 Monte Carlo runs of 0.367.

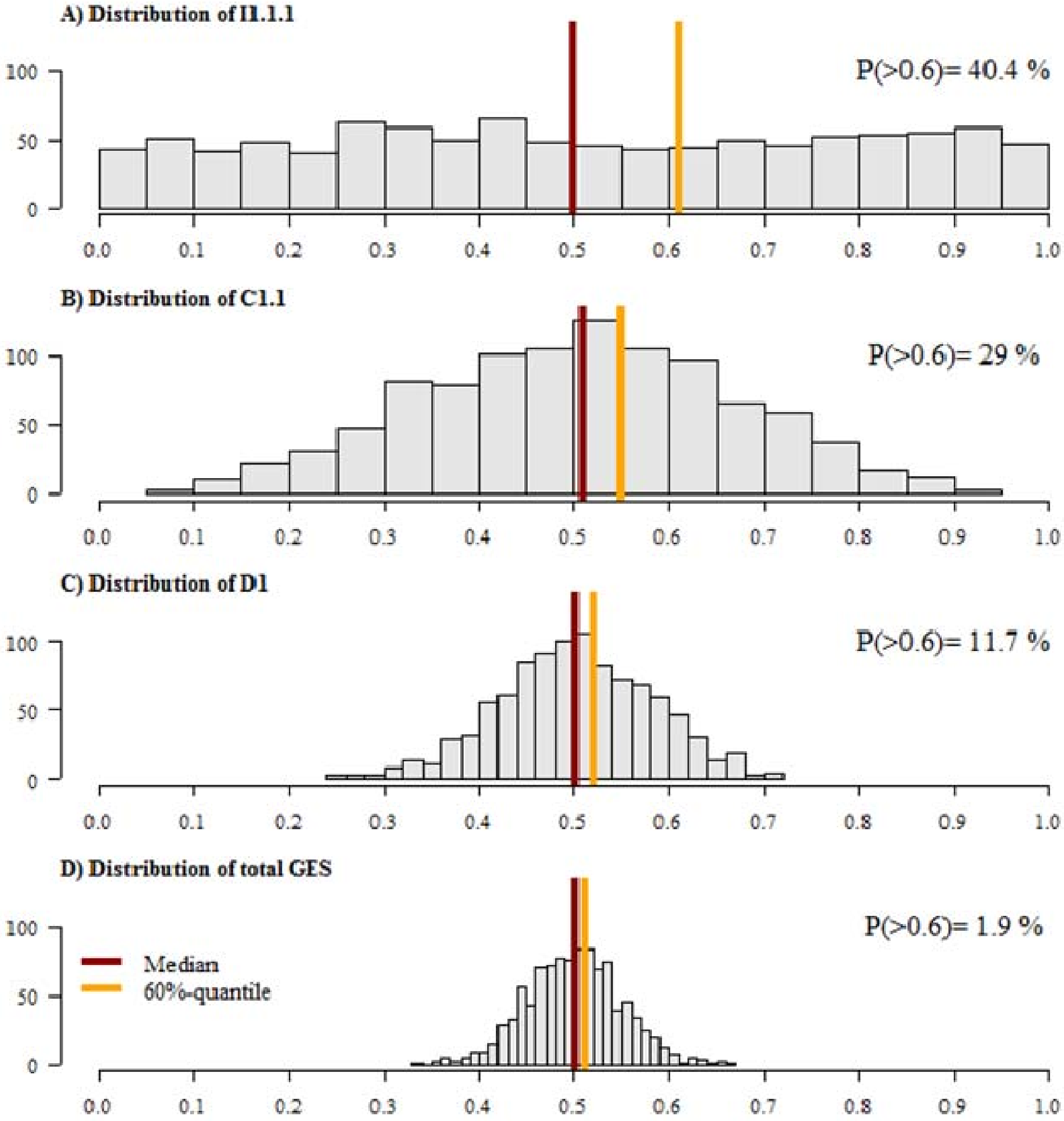


Figure 18. Implications of the Central Limit Theorem on the aggregation by AA and the ME approach. Probabilities of achieving a value >0.6 derived from 1000 Monte Carlo runs are shown for A) MSFD Indicator 1.1.1, B) Criterion 1.1, C) Descriptor 1 and D) total GES.

These results suggest that, if using either he OOAO, the AA or the ME integration method, by the time that integration at all three hierarchical levels has been completed, the probability of achieving an overall GES outcome is extremely low: approximately 0%, 1.9% and 6% respectively in 1000 trials (Figure 19: A to C). For the PROB approach, the likelihood of P(GES_{TOT}) was 44%, which was close to P(GES_I) = 0.4 (Figure 19D).

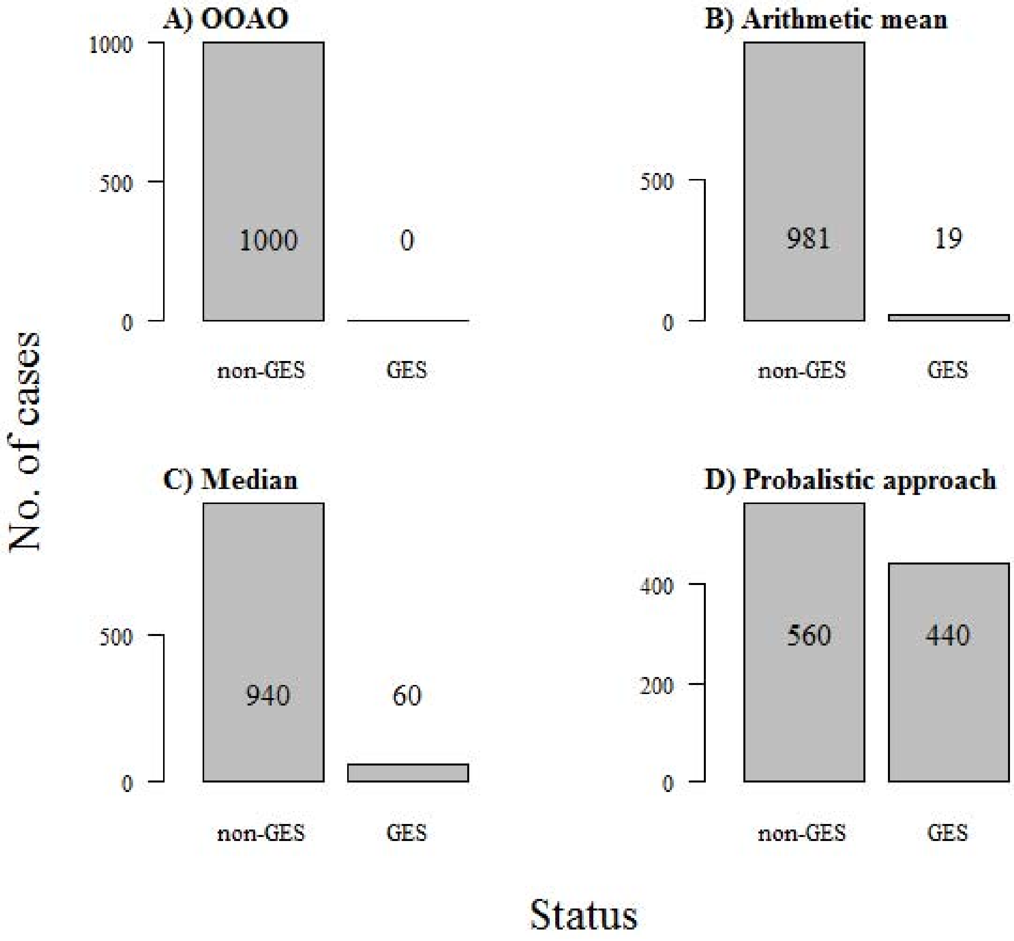


Figure 19. Probability of achieving total GES (P(GES_{TOT})) according to 1000 Monte Carlo runs for four different integration methods.

A possible solution

With the exception of the Probabilistic method, achieving GES became less likely with each increment in integration level. At Indicator level the probability of achieving GES is 0.4 using all integration methods but, if using the OOAO, AA and ME integration approaches, at Criterion level and then again at Descriptor level the probability of achieving GES decreases with each step up in the level of integration, and the extent of this decrease is dependent on the number of Indicators involved in each integration process. However, if using the Probabilistic method, the probability of achieving GES increases. From a statistical perspective, however, it may be desirable to maintain a constant probability of achieving GES at each hierarchical integration level (e.g. of 0.4 in this study): a principle of equal probability (PEP), but this could not be achieved with any of the integration methods simulated. By taking account of the Central Limit Theorem, however, the PEP could be achieved by adapting the averaging approach. To maintain the probability of GES at 0.4 within each Indicator and Criteria cluster, the upper 60%ile from each probability distribution within each cluster had to be determined. We used the 1000 Monte Carlo runs to obtain these percentiles for all Criteria (Table 10) and all Descriptors

(Table 11). For example, within Criterion 1.1, $GES_{C1.1}$ is achieved if the average score of all indicators (1.1.1, 1.1.2 and 1.1.3) is >0.55 (Table 10). For Descriptor 3, GES_{D3} is achieved if the average score for all Criteria within D3 (3.1, 3.2 and 3.3) is >0.53. At successive integration levels, the 60%ile approaches 0.5 because the normal distribution becomes increasingly narrow in line with the Central Limit Theorem (Figure 19).

Table 10. 60%iles for each MSFD Criterion obtained from 1000 Monte Carlo runs. Differences in the 60%iles are due to differences in the number of Indicators requiring integration at Criterion level for each Criterion. Criteria with a similar number of Indicators have similar 60%ile values (e.g. C1.1 and C1.3).

Criterion	60%ile	Criterion	60%ile	Criterion	60%ile
C1.1	0.55	C3.2	0.55	C7.1	0.61
C1.2	0.61	C3.3	0.55	C7.2	0.57
C1.3	0.55	C4.1	0.59	C8.1	0.60
C1.4	0.56	C4.2	0.61	C8.2	0.53
C1.5	0.55	C4.3	0.60	C9.1	0.55
C1.6	0.54	C5.1	0.56	C10.1	0.54
C1.7	0.60	C5.2	0.53	C10.2	0.61
C2.1	0.59	C5.3	0.56	C11.1	0.61
C2.2	0.56	C6.1	0.55	C11.2	0.61
C3.1	0.56	C6.2	0.53		

Table 11. 60%iles for each MSFD Descriptor obtained from 1000 Monte Carlo runs. Differences are due to variation in the number of Criteria requiring integration at Descriptor level for each Descriptor, and the number of Indicators per Criterion requiring integration at Criterion level (see Table 10). Descriptors with a similar number of Indicators involved have similar 60%ile values.

Descriptor	60%ile	Descriptor	60%ile	Descriptor	60%ile
D1	0.52	D5	0.53	D9	0.55
D2	0.55	D6	0.53	D10	0.55
D3	0.53	D7	0.56	D11	0.55
D4	0.55	D8	0.54		

By adjusting the 60%iles of the distributions observed at the Criterion and Descriptor integration levels, a consistent probability of achieving P(GES) can be maintained at each integration level, thereby achieving the PEP (Figure 20). Minor deviations from the 0.4-benchmark are simply due to stochastic variation within the 1000 Monte Carlo runs. The adjusted arithmetic mean method of aggregation may therefore provide a consistent approach to achieve P(GES) with a probability of 0.4 at all integration levels of the MSFD.

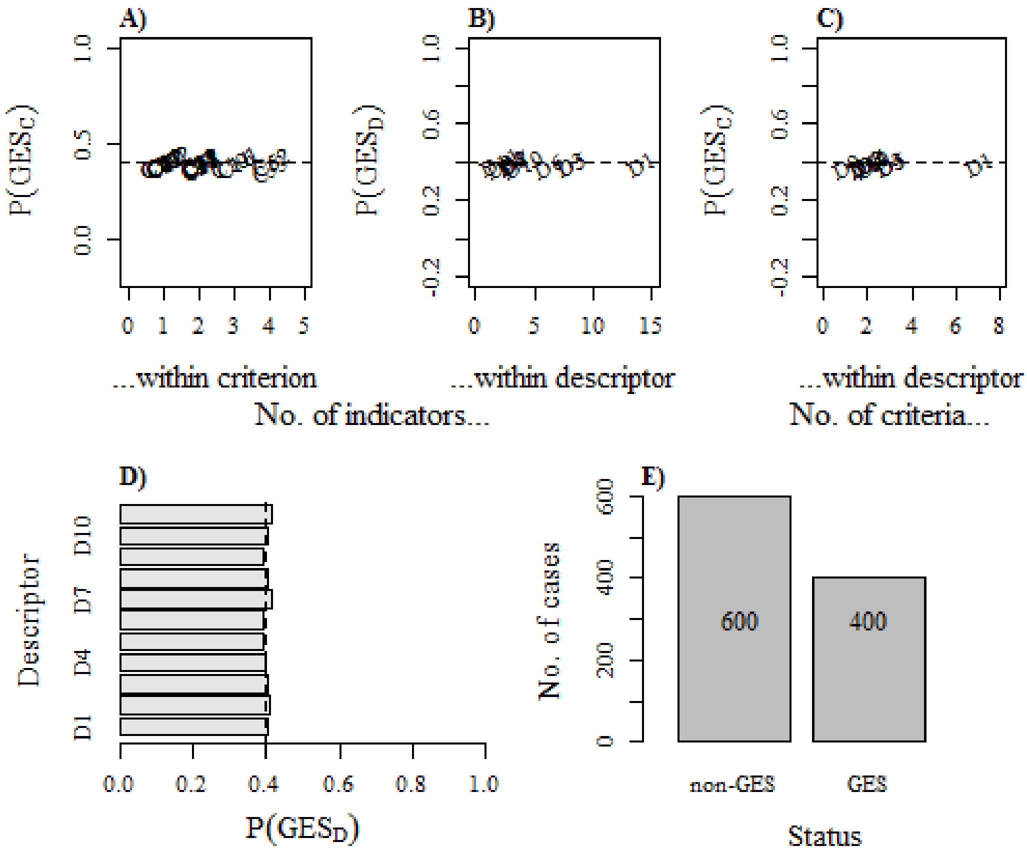


Figure 20. A suggested solution for achieving the principle of equal probability (PEP) for integrations across Indicators, Criteria and Descriptors towards GES_{TOT}. The integrations were performed using the adjusted arithmetic mean method. A) P(GES_C) and P(GES_D) independent from the number of Indicators within the Criteria and B) Descriptors as well as C) from the number of Criteria within a Descriptor. D) Thus at the Descriptor level P(GES_D) is equal to 0.4 for all descriptors alike. E) P(GES_{TOT}) is also equal to 0.4, thus PEP is maintained across all hierarchical levels within the MSFD when applying the adjusted arithmetic mean method.

Concluding comments

Integrating the information contained across multiple elements to derive a single parameter comes at a cost. Using the OOAO method ensures that overall GES at the Descriptor integration level cannot be achieved if any single Indicator involved in the assessment process falls below its GES target. If this happens, this produces a failing-GES score for the Criterion concerned at the Criterion level integration, and this failing-GES Criterion subsequently causes a failing-GES for the Descriptor involved at the Descriptor level integration. When using the OOAO integration approach, failure of a single Indicator to meet its target will eventually result in an overall failing GES outcome. The arithmetic mean, adjusted mean, median, and probabilistic integration methods permit some Indicators, Criteria or Descriptors to fail their GES whilst still delivering an overall meeting-GES assessment (Figures 19 and 20). Borja *et al.* (2014) suggests that a combined approach of OOAO, conditional rules and averaging approaches could be applied to overcome the dilemma between maintaining a constant probability for GES and allowing for the failure of single elements. The number of such combinations possible is too large to be analysed in detail in this simulation, but it is reasonable to assume that the introduction of conditional rules at any level of integration will distort the principle of equal probabilities. An example is provided in Figure 21, in which the adjusted averaging approach was combined with a conditional rule that each criterion had to be at least >0.4. The inclusion of this conditional rule lead to a distortion of the PEP reducing P(GES$_D$) below 0.4 for all descriptors.

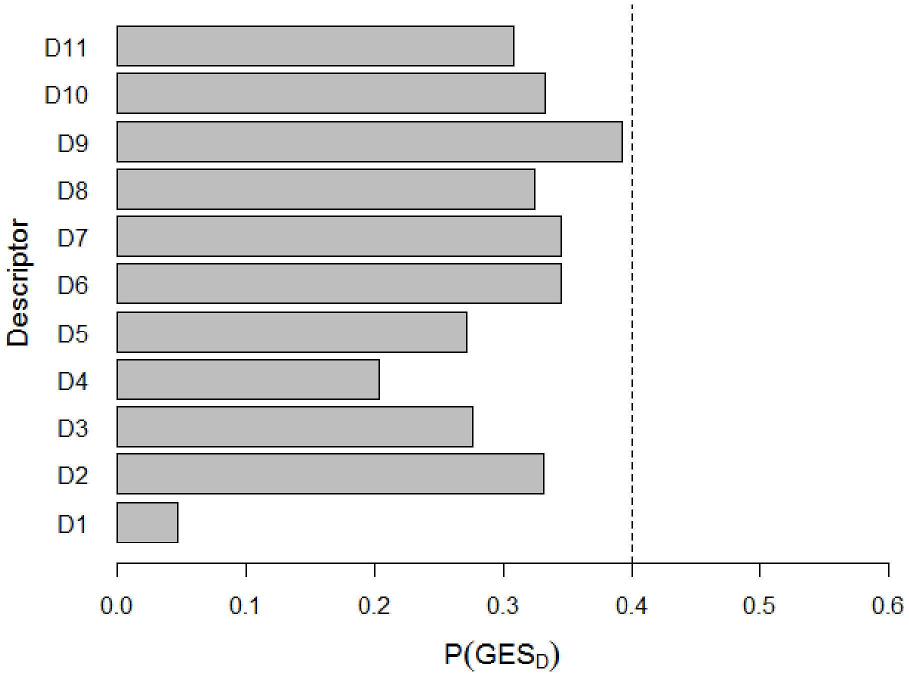


Figure 21. Example of integrated assessment outcomes using adjusted averaging integration approach and application of an additional conditional rule that all criteria must be at least >0.4 to achieve P(GES$_D$). Note how the conditional rule causes a deviation from the 0.4 benchmark by comparison to Figure 20D.

The results of this study show that integration rules have to be designed with great caution to account for statistical properties emerging from unbalanced Indicator and Criteria clusters which are structured in a nested hierarchical way. Criteria and Descriptors with few indicators will achieve GES with a different likelihood than Criteria and Descriptors with many Indicators simply because of probabilistic reasons. This study proposes a solution to this problem by suggesting individual benchmarks to each Criterion and Descriptor, which depends on the number of elements contained.

Addressing the cumulative effects of multiple pressures

The term "cumulative" is defined in the Oxford Dictionary as "increasing, increased, or formed by successive additions". A single pressure can have a cumulative effect. A constant unchanging level of fishing activity causes daily attrition in the abundance of different species. This attrition rate might vary between species but because the activity remains constant, attrition rates remain constant within species. The most sensitive species, the species whose population is most susceptible to fishing mortality and which has the least reproductive capacity to counter it, will decline in abundance at a faster rate than less sensitive species. This species will be the first to go extinct (Le Quesne and Jennings, 2012). Then the next most sensitive species will be lost from the community, and then the next most sensitive species, and so on. Thus a constant level of fishing activity, imparting constant levels of fishing pressure on different species, will have a "cumulative" effect on community species richness over time, causing successive species to go extinct. This sort of cumulative impact has been addressed in single pressure-state scenarios; here we consider cumulative impacts in the sense of the successive addition of multiple pressures associated with a variety of different anthropogenic activities.

Europe's marine ecosystems are heavily affected by many human activities that interact with a dynamic ecosystem (Halpern *et al.*, 2008a). There is increasing concern that these multiple activities interact cumulatively to accelerate biodiversity loss. For example, a variety of anthropogenic stressors, such as fishing, shellfish harvesting, nutrient enrichment, disposal of dredged sediment, have interacted to accelerate the loss of canopy-forming algae to mat-forming algae, altering the structure of many coastal plant communities globally (Lotze and Milewski, 2004; Strain *et al.*, 2014). Cumulative threat models incorporating six pressure categories, coastal-based impacts, trawling and dredging disturbance, ocean-based pollution, exploitation of marine resources by fisheries, maritime activities and impact of climate change, identified several areas of high potential risk for marine biodiversity in the Mediterranean Sea. These pressures were widespread and often overlapped with regions of high biodiversity (Coll *et al.* 2012). MacDonald (2000) defines cumulative effects as resulting from single or multiple activities in space that also persist in time. Cumulative effects can either be additive, synergistic or antagonistic. Additive effects are equal to the sum of each individual impact, while synergistic and antagonistic effects are respectively either greater than or less than the sum of the individual impacts.

Review of cumulative impact studies

In a synthesis of 171 laboratory studies that manipulated two or more stressors in marine and coastal systems, Crain *et al.* (2008) found 26% to be additive, 36% to be synergistic, and 38% to be antagonistic. Across all studies, the overall interaction effect was synergistic, but interaction type varied by response level, trophic level, and the types of stressor involved. Addition of a third stressor tended to alter interaction effects, suggesting that synergies may be common in nature where more than two stressors often coexist. Modelling the cumulative effects of fishing and acidification on the biomass and diversity of 61 functional groups or species in a southeast Australian marine ecosystem, suggested that when ocean acidification was moderate, 30% of interactions were additive, 33% synergistic and 37% antagonistic. Stronger ocean acidification reduced the frequency of additive interactions to 22% and increased the frequency of synergistic interactions to 40%. The frequency of antagonistic interactions, at 38%, was more or less unchanged. Effects were most noticeable in the demersal food web, with fishing impacting predation and acidification affecting benthic production. The model also suggested that ocean acidification and long-term fisheries exploitation could act synergistically with increasing sensitivity to change from long-term fisheries exploitation potentially causing unexpected restructuring of both pelagic and demersal food webs (Griffith *et al.*, 2012).

Even when multiple individual fisheries operating within a large marine ecosystem are well managed, the cumulative effects of all the fisheries are likely to be greater than the summed effects of each individual fleet (Rosenberg and McLeod, 2005). Individual fishery management plans aimed at achieving maximum sustainable yield for each fishery often ignore predator-prey interactions, such that if one fishery exploits an important forage fish, the productivity of its predators might be diminished thereby reducing potential yields to fisheries targeting these predators. Furthermore, they suggest that the cumulative effect on seabed habitat caused by different types of activities linked to different sectors might well be more detrimental to ecosystem services than the summed effects arising from each activity individually. For example, the impact from fishing gear in a well-managed fishery might be considered acceptable, likewise the effect of increased sedimentation from coastal development. If considered independently, the summed impacts from both activities might still be considered sustainable. However, should the combined effect involve a synergistic interaction, then the cumulative effects of both activities could instead be significantly detrimental.

Some studies have focused on the cumulative effects of anthropogenic pressures in combination with changes in environmental drivers of change in biological communities. For example, eutrophication can have a cumulative effect on zooplankton community dynamics in combination with changes in water temperature, salinity and wind conditions (Kotta *et al.*, 2009). Heavy metal pollutants, such as cadmium, cause progressive hypoxemia in bivalves, such as oysters (*Crassostrea virginica*), by impairing ventilation and circulation. This effect is more marked at higher water temperatures, suggesting a synergistic cumulative effect whereby the thermal tolerance window is narrowed in bivalves inhabiting polluted areas (Lanning *et al.*, 2008). In the early life stages of two bivalves *Argopecten irradians* and *Mercenaria mercenaria*, the stressors of low oxygen concentration and increased acidification reduced both growth and survival rates more when combined than would have been expected by either stressor individually; the cumulative impact was additive, being equal to the sum of the two individual effects. In later life-stages, however, both species became relatively resistant to each stressor individually, but in a synergistic response, suffered markedly reduced growth rates when subjected to both stressors in combination (Gobler *et al.*, 2014). There is some concern that human pressures, such as the introduction of heavy metal pollutants or nutrients into marine ecosystems, could exacerbate the effects of climate change on plankton and coral communities (Rose *et al.*, 2009; Russell *et al.*, 2009).

Changing environmental conditions can strongly influence recruitment rates in fish populations, such that the cumulative effects of changing environmental drivers combined with overfishing are likely to be synergistic. The decline of cod in the North Sea provides evidence of this. This stock was able to sustain high levels of fishing mortality during the 1970s and into the 1980s when water temperatures were relatively cool, but despite reducing levels of fishing pressure, the stock declined markedly through the late 1980s and

1990s as water temperatures rose (Kirby *et al.*, 2009). In a study of the additive, synergistic and antagonistic cumulative effects of fishing, ocean warming, and ocean acidification on the biomass of five community groups, top predators, fishes, benthic invertebrates, plankton, and primary producers, comprising 60 functional groups or species, in a south-eastern Australian marine ecosystem, only ocean acidification had a negative effect when considered in isolation. Both combinations of fishing and ocean warming, and ocean warming with ocean acidification, had additive effects. However, adding fishing to the ocean warming and ocean acidification combination significantly changed the direction and magnitude of the interaction to one of a synergistic negative response on biomass (Griffith *et al.*, 2012). In Kenyan coral reefs, fishing reduced coral cover by 51% and bleaching associated with increased water temperature reduced coral cover by 74%. However, the two stressors combined were only weakly additive, or perhaps even antagonistic, possibly because bleaching was the dominant stressor such that when temperature related bleaching occurred, the additional effect of fishing was minimal (Darling *et al.* 2010).

The prevalence and magnitude of these cumulative effects, and their influence on future ecological change, remains largely unknown (Darling and Côté 2008); where, when, and to what degree pressures interact in non-additive ways, and the extent to which these interactions vary over time. Management decisions regarding which activities require attention are particularly vulnerable to this knowledge gap. Reducing an antagonistic pressure could make conditions worse, whereas reducing a synergistic pressure could produce much greater benefit than would arise from similar attention to an additive pressure (Halpern and Fujita, 2013). For example, in a meta-analysis of 118 studies of the impact of anthropogenic pressures on the loss of canopy-forming algae to matt-forming algae, many of the cumulative impacts were additive, but those synergistic interactions observed tended to involve nutrient enrichment as one of the pair of pressures. Management to control nutrient enrichment might therefore have the greatest beneficial effect of reducing the loss of canopy-forming algal biodiversity (Strain *et al.*, 2014).

Implications for integrated ecosystem assessment

IEA intended to support EBM has to take account of the cumulative effects of the different activities associated with multiple sectors operating in the marine environment (Lotze, 2004; Browman and Stergiou, 2005; Hirshfield, 2005; Halpern *et al.*, 2008b; Ban *et al.*, 2010; Katsanevakis *et al.*, 2001). Estimates of the risks arising from these multiple human pressures also need to be determined at meaningful ecological scales (Eastwood *et al.*, 2007; Halpern *et al.*, 2008b; Fock *et al.*, 2011). The MSFD acknowledges this by calling for a holistic assessment of pressures on marine ecosystems. Article 1(3) states that "*Marine strategies shall apply an ecosystem-based approach to the management of human activities, ensuring that the collective pressure of such activities is kept within levels compatible with the achievement of good environmental status* …". Article 8(1) goes on to state therefore that assessments of status should include "*an analysis of the predominant pressures and impacts, including human activity,* …", consider "… *the qualitative and quantitative mix of the various pressures,* …" and take account of "… *the main cumulative and synergetic effects;* …" (EU, 2008). Consequently, the last five years have witnessed a proliferation of efforts to characterise and map cumulative impacts, providing insight into the strengths and limitations of these efforts and where opportunities for progress lie (Figure 22).

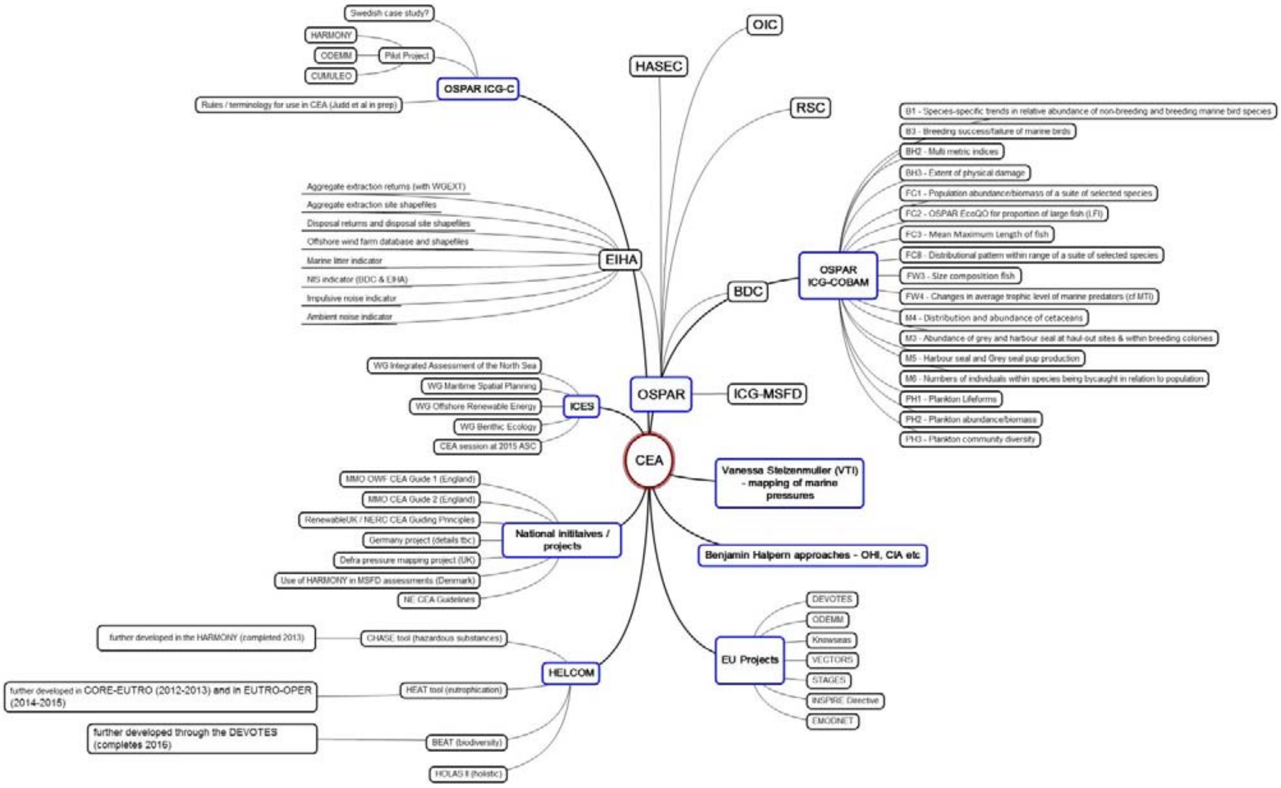


Figure 22. (A Judd, Cefas). Various international work teams associated with cumulative effects assessments (CEA). Note that there are a number of ICES Working Groups that have CEA in their ToRs).

Conceptual models for multiple activities and pressures

Given the variety of approaches proposed for addressing single pressure-state relationships, it is not surprising that currently there is little consensus on methods to identify, predict and assess the effects of cumulative pressures on marine ecosystems. Provision of an analytical framework for evaluating cumulative effects of human activities on a suite of ecosystem services is therefore a key prerequisite to successful integrated EBM (Rosenberg and McLeod, 2005; Tudela and Short, 2005).

With respect to managing multiple human impacts on marine habitats, a mapping approach has shown some promise. Spatial variation in the intensity of each pressure is mapped and overlaid on a map of the habitat in question. A vulnerability weighting is then assigned to each pressure that translates pressure intensity to estimated impact on the habitat, thereby creating a single pressure impact ‘currency’ that facilitates a summing procedure to derive a total cumulative impact score for all pressures acting on the habitat (Halpern *et al.*, 2008a; Korpinen *et al.*, 2012). However, this approach has limitations: it assumes that mapped pressure layers are of equal importance; that habitats respond linearly to increases in the intensity of a pressure; that all locations across the mapped habitat respond to each pressure in the same way; that cumulative interactions are unaffected by variations in the ratios of different pressures acting at each habitat location; and that quantitative vulnerability weightings for each habitat-pressure combination are accurate (Halpern and Fujita, 2013).

The OSPAR Inter-sessional Correspondence Group on Cumulative Effects recently reviewed cumulative effects assessment (CEA) methodologies adopted in three different European projects (Korpinen 2014), namely:

1) the CUMULEO method of IMARES (The Netherlands);
2) the method developed within the FP7 project ODEMM (project co-ordinator University of Liverpool, UK); and
3) the method used in the HARMONY project based on Halpern (2008) (project co-ordinator Aarhus University, Denmark).

Although certain differences do exist, the basic underlying approach in all three methods was similar and relied on mapping GIS data on pressures (either presence/absence or extend of habitat loss) and the ecosystem components (presence/absence or species biodiversity). An impact score was then calculated based on the level of impact and recovery time, which was subsequently multiplied with pressure and ecosystem components to derive a measure of impact (and recovery) per grid cell. A major common limitation was the lack of a systematic process for selecting data layers of pressures and ecosystem components. Such a process is essential for balanced selection of different kinds of pressures in the assessment. All three projects considered only additive models although these can be adapted to accommodate synergistic or antagonistic effects, but the issue here is recognising when and to what extent non-additive cumulative effects occur.

The assumptions underpinning this mapping approach cannot always be met and an alternative view is that an indicator-based management framework to address the cumulative impacts associated with multiple human pressures acting on marine ecosystem components will emerge from the evolution of management frameworks, such as the PSR, APSR, and DPSIR, that have been applied to manage the effects of single pressures individually. Because of the variety of possible individual activity-pressure-state scenarios, no single solution to the more complex issue of undertaking integrated assessments of multiple human activities and pressures and addressing multiple management objectives is obvious (Apitz *et al.*, 2006). Within a multiple activity – multiple pressure situation, the APSR framework, for example, could have several different representations (Figure 23). Knights *et al.* (2013) used simple A-P-S-R linkages (which they termed "impact chains") to build an integrated network. This captures a diverse range of sector activities that impart a variety of pressures that interact to cause a change in state, the impact, in an ecological characteristic (e.g. MSFD indicator), which in turn requires a management response Figure 24). Knights *et al.* (2013) expanded the 18 pressures listed in Annex III of the MSFD (see Table 4) to 21, adding, for example, changes in wave exposure and their effects on intertidal communities, pressures that could arise from new but increasingly prevalent activities such as barriers to species movement arising from renewable-energy installations. Of the 18 ecological characteristics of the marine environment listed in Annex III of the MSFD, Knights *et al.* (2013) considered 14 and grouped these into four broad categories: (1) physical and chemical features (i.e., temperature, salinity, topography, nutrients and oxygen, pH); (2) predominant habitat types (e.g., sublittoral sediment, deep sea, littoral rock); (3) biotic characteristics (fish, birds, mammals, benthic flora and fauna, plankton, listed species), and (4) other notable chemical features of the ecosystem (e.g., presence or absence of hotspots for eutrophication).

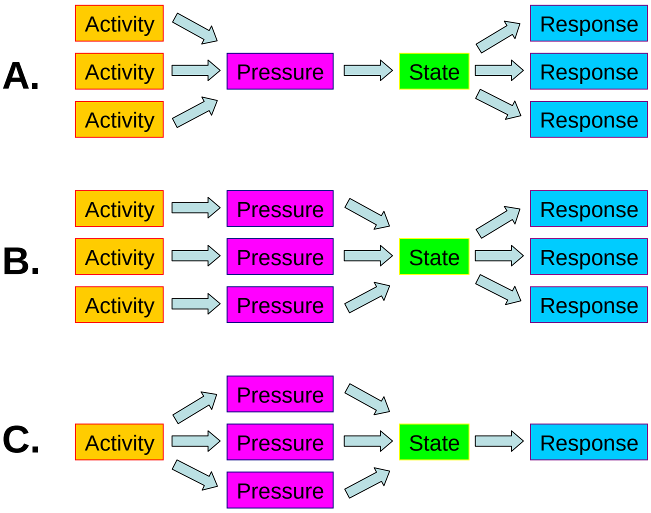


Figure 23. Representation of various potential activity-pressure interactions affecting a state indicator. A) Three separate "activities" (e.g. fishing, gravel extraction, and waste dumping) combine to cause a single "pressure" (e.g. mortality of a benthic invertebrate), which alters the "state" (e.g. population biomass) of the ecosystem component monitored. B) Three separate "activities" (e.g. fishing, gravel extraction, and waste dumping) cause three different "pressures" (e.g. physical damage, habitat loss, and smothering. C) A single "activity" (e.g. fishing) causes three separate "pressures" (e.g. mortality, physical damage, habitat loss), which combine to alter the "state" (e.g. population biomass) of the ecosystem component monitored. Note that in each case separate management "responses" are necessary to regulate each activity.

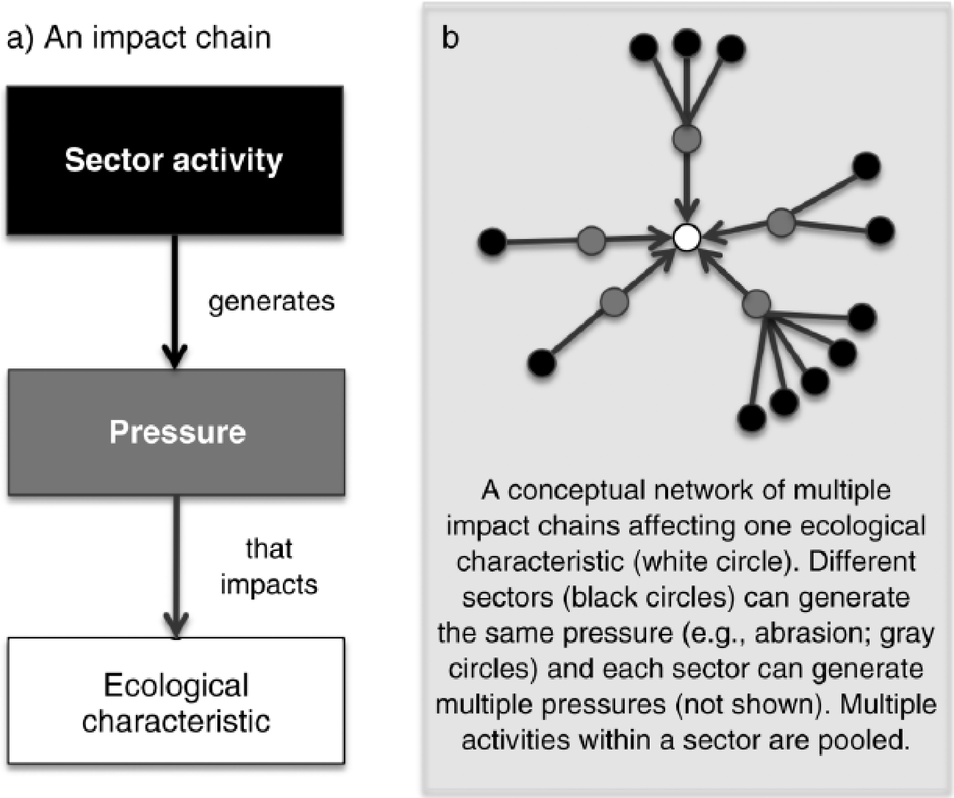


Figure 24. Impact chains following Knights *et al.* (2013). (a) A generic hierarchical impact chain linking sectors and activities to an ecological characteristic via a specific pressure. A sector is a distinct industry, such as fishing or shipping that undertakes specific activities (e.g., benthic trawling) during its exploitation of marine resources. Each sector activity can generate many different pressures that impact one or more ecological characteristics or cause harm to the environment (e.g. benthic trawling extracts fish species). An ecological characteristic (a habitat, species, or interest feature of an ecosystem, e.g., demersal fish) can be impacted by multiple sectors and multiple pressures, forming (b) a complex network of sector–pressure impact chains. A separate impact chain is generated for every combination of sector (black circles), pressure (gray circles), and ecological characteristic (central white circle).

Bow-tie diagrams can also be used to describe and analyse risk events and risk management options by visualising relevant pathways from causes to consequences (Cormier *et al.*, 2013; Ferdous *et al.*, 2013). The elements of a bowtie have a specific terminology but can be adapted to fit the DPSIR or APSR frameworks as presented in Figure 25. At the centre of the bowtie is the so-called "top-event", which describes the loss of control over the hazard or the risk source and corresponds to an undesirable "state change". The main difference of a bow-tie to DPSIR or APSR is that it was mainly developed to evaluate the effectiveness of existing or potential management measures to reduce the likelihood of a top event occurring or to mitigate the consequences resulting from a top event. By itself, it does not quantify pressure-state-relationships *per se*. In a bow-tie, the response is placed either as prevention "controls" between threats or sources of risk (driver, activity), causes (pressures) and the top event (state) or as mitigation barriers between the top event and the consequences (impacts). Bow-tie analysis has been introduced in various environmental management contexts including cumulative effects assessments. Multiple bow-ties can also be chained together to integrate cumulative assessments at multiple management scales.

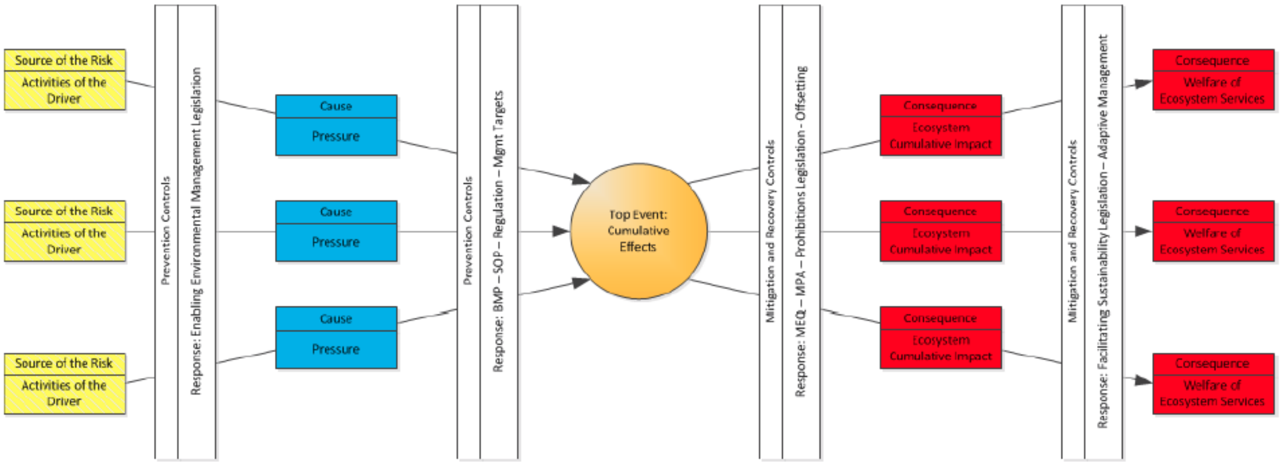


Figure 25. Bow-tie conceptual framework adapted from Cormier *et al.* (2013).

Bayesian belief network (BBN) analysis is another emerging assessment methodology that goes beyond the basic mapping approach. BBNs start out with a conceptual model representing the components under analysis as nodes that are connected via acyclic links. The conceptual model can be built in such a way that it represents any form of an APSR/DIPSIR model. Recently, bow-tie diagrams have been used in combination with BBNs to overcome their purely depictive capabilities by adding probabilities and conditional dependencies between components (Badreddine and Amor, 2013; Khakzad *et al.*, 2013). In addition, the BBN allows for the quantification of uncertainty that a bow-tie on its own cannot. For the analytical part, each BBN node is assigned a conditional probability table which defines the links from each node by specific mathematical relationships. However, this is not trivial and has currently not been done in environmental assessments. Further development of this approach is therefore needed. Stelzenmüller *et al.* (2014) used additive effects to analyse the effects of six different fishing fleets on benthic organisms. A BBN can be fed with data and expert knowledge and it can also be implemented within a GIS to make spatially explicit assessments.

Concluding comments

Whether or not analysis of cumulative effects can be developed into being an operational component of IEA to support EBM implementation under the MSFD depends on two principal factors. Firstly, while many studies have empirically tested cumulative relationships in controlled experiments, for example in microbial studies, uncertainties still remain with regard to large-scale real-world systems where multiple activities take place. The second issue relates to the added layer of complexity in determining appropriate management responses in situations where deterioration in a state indicator cannot be traced back to a single activity or sector. Addressing all changes in ecosystems is a complex task. Establishing causal relationships between pressures and observed impacts in natural systems is difficult due to the presence of other biotic and abiotic factors that can also simultaneously modify ecosystem responses to pressures (Frederiksen *et al.*, 2004), the existence time lags between changes in pressure and the biological response (Greenstreet *et al.*, 2011; Probst *et al.*, 2012), and the possibility of multiple pathways by which pressures can affect ecosystem structure and function. The complexity of marine ecosystems, their high variability and the broad array of activities that impact many aspects of these ecosystems suggests that no single measure may be adequate for addressing the effects of multiple pressures (Boldt *et al.*, 2014).

Crain's *et al.* (2008) meta-analysis of 171 studies that manipulated two or more pressures in marine and coastal ecosystems underlines this point. The type of interaction, additive, synergistic or antagonistic, varied by:

1) Response level (community versus population): Species-level impacts can be reduced or exacerbated through species interactions. Species interactions within communities dampen and diffuse the impacts of multiple pressures that can have strong negative effects on a given species. Consequently, species-level MSFD indicators may have limited utility in predicting community or ecosystem response to multiple pressures.

2) Trophic level: Trophic level is likely to drive interactions between pressures because organisms with fundamentally different methods of energy acquisition may respond differently to pressures (e.g. increased CO_2 can benefit plants but harm invertebrates due to increased ocean acidification). Interactive effects of multiple pressures could be more negative for organisms at higher trophic levels, possibly due to the loss of biological insurance as taxonomic, physiological and genetic diversity decrease with increasing trophic level (Christensen *et al.*, 2006).

3) Pressure combination: As the number of pressures in a system increases, pressure interactions become increasingly complex. Understanding the mechanisms by which each activity or pressure individually drives population or community response helps to interpret or predict when and where pressures interact. For example, two pressures that affect population size directly (overexploitation and immigration due to habitat patchiness) have been shown to have additive effects (Mora *et al.*, 2007). However, when these pressures were each paired with temperature increases due to global warming, the pairwise interactions were both synergistic. Because response to warming depends to some extent on genetic diversity, decreasing population size (i.e. due to overexploitation) may disproportionately affect the response to warming. Thus, pressures acting through similar mechanisms may be additive, while those acting through alternative but dependent pathways may be synergistic.

For the exploration of policy and management options it is helpful to distinguish between activities that are directly amenable to management (e.g. fishery in a catchment) and drivers of ecosystem change that cannot be influenced directly (e.g. climate change, storm events, etc.). EBM is thus attempting to compensate for the impacts of global drivers by reducing local pressures, such as overfishing. This approach assumes that pressures interact additively or synergistically, but it is not clear from this whether management need react to antagonistic interactions among pressures, and if so how.

Brown *et al.* (2013) examined the effectiveness of management when faced with different types of pressure interactions in two systems, seagrass and fish communities; climate change constituted the global pressure but the local pressures differed. Simple models were used to illustrate how global climate and local pressures can interact to influence the type of management measures needed to maintain populations and communities. When there were synergisms, mitigating local pressures delivered greater gains, whereas when there were antagonisms, management of local pressures was ineffective or even degraded ecosystems. These results suggest that reducing a local pressure can compensate for climate change impacts if there is a synergistic interaction. Conversely, if there is an antagonistic interaction, management of local pressures will have the greatest benefits in areas of refuge from climate change. There was little that local management could do to counter severe impacts of climate change on populations and communities. In such cases, revision of MSFD targets may be necessary.

In summary, successful EBM will require assessment of the cumulative effects of multiple pressures on ecosystem components as well as the development of adequate scientific knowledge in order to be in a position to provide appropriate advice as to how to address these cumulative impacts. While various conceptual frameworks exist, progress needs to be made to advance the actual assessments of cumulative effects of multiple pressures. In the absence of clear knowledge when and to what degree synergistic effects take place, the default model will remain the additive model. Advances need to be made on how to translate results from assessment to management advice. From a management perspective, the challenge lies in identifying environmental management priorities that address the most significant pressures and ecosystem vulnerabilities.

Compatibility between targets representing GES for different indicators

Ideally, EBM should be holistic in nature, i.e. attempting to manage towards some overall condition of the whole marine ecosystem. In reality, our level of knowledge is often inadequate to support such a utopian goal. Science tends to focus on understanding individual components of marine ecosystems and the various separate processes by which each focal component interacts with other ecosystem components and with its physical environment. Earlier development of an EAM by OSPAR, the North Sea EcoQO project, reflected this situation; setting individual targets (the EcoQOs) for specified indicators, which tended to quantify variation in individual aspects of specific components of marine ecosystems (Heslenfeld and Enserink, 2008). Implementation of the MSFD appears now to be following a similar route. The 2010 MSFD Decision Document lists 56 separate indicators across all eleven Descriptors; but a disproportionate fraction of these, fourteen indicators (25%), apply just to Descriptor 1 "Maintenance of biological diversity". Furthermore, when considering how to co-ordinate implementation of the MSFD between Member States (MS) sharing specified MSFD regions or sub-regions (e.g. the Greater North Sea), OSPAR proposed that metrics to fulfil the 14 D1 indicator functions should be derived for five separate ecosystem components: Birds, Mammals and Reptiles; Fish and Cephalopods, Pelagic Habitats and Benthic Habitats. Even taking account of the fact that certain MSFD indicators apply only to some ecosystem components and not to others, this approach will involve the use of many individual metrics, each with its own target representing GES.

With a multitude of individual metrics fulfilling 14 indicator functions across five separate ecosystem components, it seems inevitable that targets set for any particular metric, which might seem entirely rational when the target for each specific metric is considered in isolation, might in fact not be so sensible when targets for all the metrics, indicators and ecosystem components are considered as a single set. Targets set for one metric may fail to take account of the full complexity of the food web interactions linking all the ecosystem components, and may simply be unattainable should the targets for other MSFD indicators be achieved. Inconsistency in selection of reference periods to inform target setting may well result in the setting of mutually incompatible targets purporting to represent GES for the different indicators used to monitor change and assess the status of different components of the marine ecosystem. For example, targets relating to the management of fisheries may well aim to return to some former historic state observed in the past when fishing was deemed to be at sustainable levels, whereas conservation goals for seabirds and marine mammals may well aim to preserving current status, when populations of many species are comparatively high. This could easily produce inconsistent targets whereby one or other target might be achievable, but not both. Link (2010) suggests that IEA will have to confront such trade-offs, while Sissenwine and Murawski (2004) suggest that "Ecosystem-based approaches will increasingly be viewed as a mechanism for resolving conflicting objectives arising from the species-by-species approach".

If targets are incompatible, then it is inevitable that some targets will be missed simply because others are met. In such circumstances further restricting human activity in the marine environment will not provide a solution. Instead targets will have to be adjusted so that they become complementary, and as a consequence, allow a 'meeting-GES' outcome in an IEA. However, since this could be perceived as a relaxation of management ambition, the evidence needed to support such target adjustment will need to be compelling. Our purpose in this section therefore is to start the process of assimilating such evidence should it be required. Case study examples are organised by ecosystem component and MSFD Decision indicator.

Incompatibility between fisheries management D3 and seabird D1 and D4 targets

Seabird breeding success targets

Three separate indicators address the breeding success of seabirds. B-2, the 'annual breeding success of kittiwakes', and B-3, the 'breeding success/failure of marine birds', are Descriptor 1 indicators supporting the requirement to maintain biological diversity, and FW-1, the 'reproductive success of marine birds in relation to food availability', is a Descriptor 4 indicator addressing the need to maintain food web structure and function (OSPAR, 2013b). Reproductive rate is in important process controlling seabird population dynamics (Crespin *et al.*, 2006; Sandvik *et al.*, 2012); ensuring adequate breeding success is therefore essential if declines in abundance of seabirds are to be avoided and seabird community biodiversity maintained. Variation in the breeding success of seabirds has frequently been linked to changes in the availability of prey (Cairns 1987; Harris and Wanless, 1997; Monaghan *et al.*, 1989; Frederiksen *et al.*, 2006; Daunt *et al.*, 2008), thus providing the link between seabird breeding success and food web processes. It was in this role that the kittiwake breeding success indicator was initially developed to support the OSPAR Ecological Quality Objectives pilot study for North Sea (Heslenfeld and Enserink, 2008). This predator-prey interaction also provides the link between these seabirds' breeding success indicators and a manageable human activity in the form of the industrial fishery for the small pelagic fish that constitute the principle prey of most seabird species in European waters (Furness and Tasker, 2000; Furness, 2002).

The target for indicator B-2, the 'annual breeding success of kittiwakes' is well developed, being defined by an established relationship with winter sea surface water temperature (Frederiksen *et al.*, 2004). This means that there is no set target; instead the target can vary from year to year reflecting recent environmental variability. In contrast, no similar procedure has been developed for indicators B-3 and FW-1. Indicator B-3 was intended to compliment B-2, to cover other seabird species and to provide a similar breeding success indicator in areas or subregions where kittiwakes were not present (OSPAR, 2013a). For both Indicator B-3 and Indicator FW-1, the target proposed at the Criterion level is "*widespread seabird colony failures (defined as the production of ≤ 0.1 chicks fledged per nest* (Cook *et al.*, 2012)) *should occur rarely in other species that are sensitive to changes in food availability*". This Criterion target would be assessed on the basis of the number of species achieving species-specific supporting targets: "*The annual percentage of colonies experiencing breeding failure does not exceed the mean percentage of colonies failing over the preceding 15 years* (appropriate for species, such as the arctic tern *Sterna paradisaea*, that naturally frequently fail to breed), *or 5%* (appropriate for species, such as guillemots *Uria aalge*, that naturally rarely fail to breed), *whichever value is greater, in more than three years out of six*".

For all three seabird breeding success indicators, the targets reflect a relatively recent situation. For B-3 and FW-1 this is explicit in that the target for the acceptable frequency of breeding failures is based on the mean of the last 15 years of the time series. But even for B-2, where the target is dependent on prevailing winter sea surface temperatures, the relationship on which this target setting procedure is based was derived from data collected towards the latter years of the 20th century and the early years of the 21st century. During this time, populations of most fish species targeted by commercial fisheries were seriously depleted following decades of over-exploitation (Hislop, 1996; Piet and Rice, 2004; Halpern *et al.*, 2008a). Many of these fish species are piscivorous, consuming exactly the same species of small pelagic fish as seabirds. Under the MSFD, targets for commercial fish stocks require reductions in fishing mortality to allow populations to increase in size towards levels that prevailed earlier on in the 20th century. Conversely, targets for these three seabird breeding success indicators require the situation and conditions prevailing towards the end of the 20th century to be conserved. It is conceivable that these two sets of targets may be mutually incompatible.

In 'wasp-waist' food webs (Cury *et al.*, 2000; Bakun, 2006; Yaragina and Dolgov, 2009; Bakun *et al.*, 2010; Schweigert *et al.*, 2010) energy from species-diverse lower trophic levels is channelled to a broad range of predators in higher trophic levels via a single, or just a few, key species that dominate the intermediate trophic level (Rice, 1995; Frederiksen *et al.*, 2006). In the North Sea, the lesser sandeel *Ammodytes marinus* (Raitt 1934) fulfils such a role, obtaining energy from a wide range of zooplankton prey (Macer, 1966; Reay, 1970; Meyer *et al.*, 1979) and in turn constituting a major component of the diets of many fish, seabird and marine mammal predators (Daan, 1989; Hislop *et al.*, 1991; Tollit *et al.*, 1997; Greenstreet *et al.*, 1998; Furness and Tasker, 2000; Santos and Pierce, 2003; Marubini *et al.*, 2009). However, while sandeels may be a key prey for seabird predators, seabirds are far from being the most important predators in terms of their capacity to influence the population dynamics of sandeels and other forage-fish species. Approximately 90% of all forage fish (including sandeels) consumed by predators in the North Sea are eaten by piscivorous fish (Bax, 1991; Mackinson and Daskalov, 2007).

Fishing pressure in the northeast Atlantic increased markedly during the 20th century (Thurstan *et al.*, 2010), causing substantial reductions in predatory fish populations (Christensen *et al.*, 2003; Myers and Worm, 2003). Such changes in predator abundance can initiate trophic cascades, causing populations of their small-fish prey to increase in abundance by as much as an order of magnitude (Worm and Myers, 2003; Ward and Myers, 2005; Frank *et al.*, 2005; Frank *et al.*, 2007; Myers *et al.*, 2007; Heithaus *et al.*, 2008). In the North Sea, where prolonged heavy fishing has reduced predatory-fish populations (Hislop, 1996), increases in small prey-sized fish biomass have exceeded the decline in predatory-sized fish biomass by a factor of at least five (Sherman *et al.*, 1981; Daan *et al.*, 2005; Greenstreet *et al.*, 2011). Over the same period, numbers of seabirds utilising North Sea feeding grounds have increased exponentially. Increased protection from hunting, reducing 'top-down control' of seabird populations, may in part have contributed to this expansion of seabird numbers (Mitchell *et al.*, 2004), but it must also have been facilitated by the relaxation of 'bottom-up limitation' associated with increased small fish abundance (Montevecchi, 2002; Furness, 2002). By the end of the 20th century, the rate of increase in seabird numbers had slowed and some decline was even evident (Mitchell *et al.*, 2004; JNCC, 2009), suggesting that seabirds may have reached their environmental 'carrying capacity' so that populations were now becoming 'bottom-up limited' by prey availability.

Clear 'bottom-up limitation' effects on feeding behaviour and breeding performance have been described in one sandeel-dependent seabird predator in particular, the black-legged kittiwake *Rissa tridactyla* (L. 1758); both environmental and industrial fishery influences on sandeel availability to kittiwakes have been demonstrated (Harris and Wanless, 1997; Lewis *et al.*, 2001; Frederiksen *et al.*, 2004; Frederiksen *et al.*, 2005; Scott *et al.*, 2006; Frederiksen *et al.*, 2007; Daunt *et al.*, 2008). Figure 26 shows the graph published by Frederiksen *et al.* (2004), which demonstrates a clear negative relationship between sea temperature and kittiwake breeding success on the Isle of May in the Firth of Forth, Scotland, linked strongly to the capacity of adult kittiwakes to provision their chicks adequately with sandeel prey. Two relationships are shown, one for years when no sandeel fishery operated off the Firth of Forth, and the second when the sandeel fishery was active. A clear negative displacement of the relationship is apparent in years when the sandeel fishery was in operation, indicating the negative impact of the fishery on the availability of sandeel prey to adult kittiwakes foraging to feed their chicks. However, in the same Firth of Forth study area and over the same period, the population abundance and feeding performance of two of the most abundant fish predators of sandeels in the area, whiting *Merlangius merlangus* (L. 1758) and haddock *Melanogrammus aeglefinus*, have been shown to be entirely independent of variability in sandeel abundance. These two fish predators appeared to experience little in the way of 'bottom-up limitation' associated with variation in the availability of sandeel prey. Total consumption of sandeels by whiting and haddock predators was directly proportion to the population biomass of the two fish predators (Reilly *et al.*, 2014).

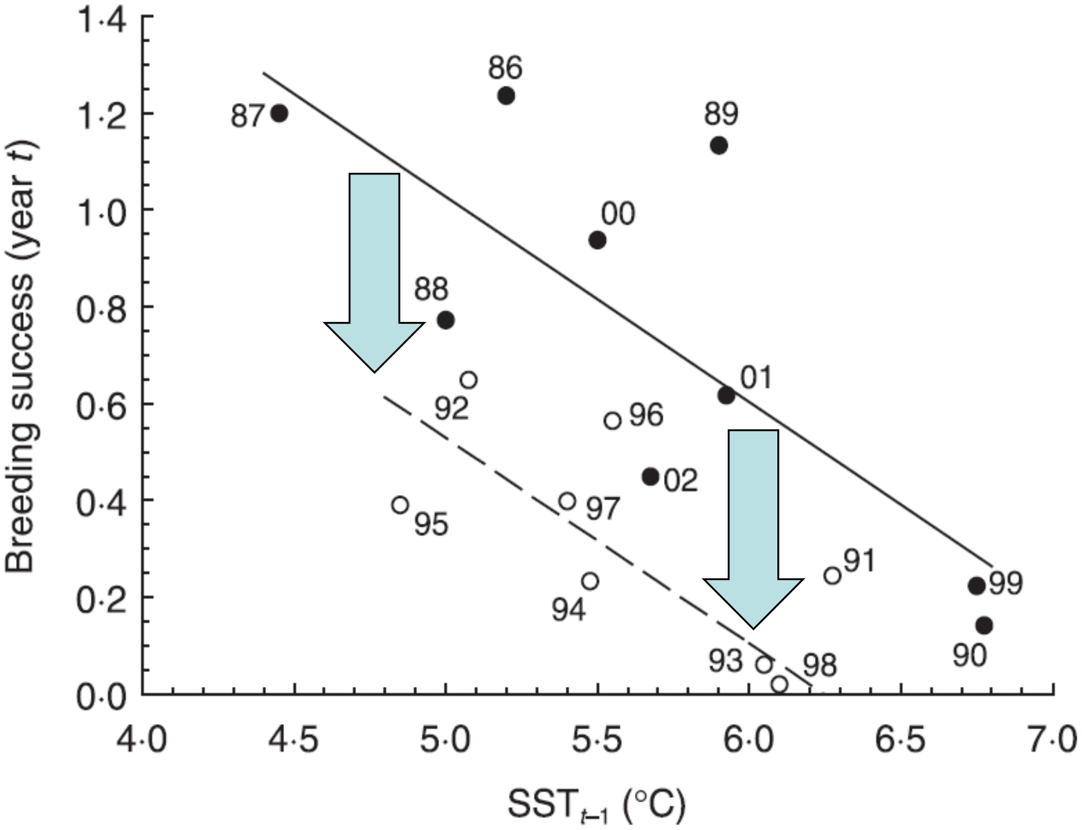


Figure 26. The relationship between kittiwake breeding success at the Isle of May, Firth of Forth, Scotland, and sea temperature during the preceding winter in years when no sandeel fishery was in operation on the nearby Wee Bankie (solid dots and line) and in years when the sandeel fishery was active (open dots and dashed line). Arrows indicate the negative displacement of the relationship in years when the fishery was active, indicating the reduction in sandeel availability to foraging adult kittiwakes caused by the fishery in these years. Modified from Frederiksen *et al.* (2004).

For a species to exist at a constant equilibrium population size, the amount of a limiting resource available must be sufficient to support a population birth rate that is equal to the population mortality rate: termed the 'equilibrium resource requirement' (Tilman, 1982). If resources exceed the 'equilibrium resource requirement' the population will increase, if lower the population will decrease. Tilman (1982) maintains that the outcome of competition for a single limiting resource by two or more species is predictable on the basis of the 'equilibrium resource requirements' of each species; the species with the lowest 'equilibrium resource requirement' wins, the other species loses and will decline in abundance. Whilst experiencing the same range in the abundance of sandeel prey, kittiwakes in the Firth of Forth were food limited while haddock and whiting were not. This is illustrated by hypothetical Holling type II feeding responses for whiting and kittiwakes (Figure 27) implied by these two sets of results (Frederikesen *et al.*, 2004; Reilly *et al.*, 2014). The two gadoid species would appear to have a lower 'equilibrium resource requirement' and should win in a competition with kittiwakes for sandeel prey.

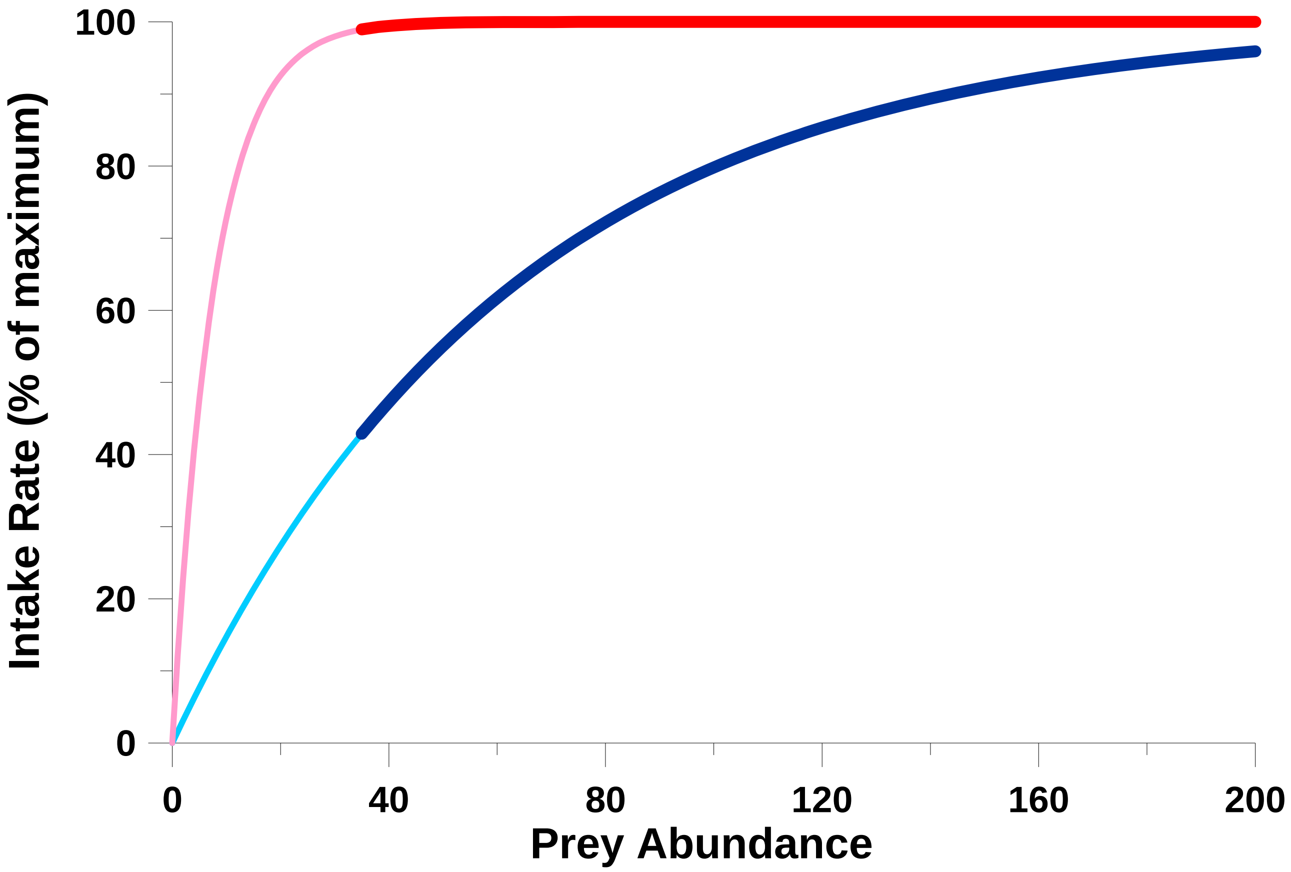


Figure 27. Hypothetical Holling type II feeding rate responses to variation sandeel prey abundance by piscivorous fish (red line) and kittiwakes (blue line). Heavy lines indicate the range of observed sandeel prey abundance, and comparison of the two plots indicates that, towards the lower end of this range, kittiwake feeding performance is diminished to a greater extent than that of piscivorous fish, implying that piscivorous fish would be the dominant competitor for sandeel prey (Tilman 1982).

If ecological and fisheries objectives for fish stocks and communities are successfully achieved, populations of haddock and whiting, both across the North Sea and within the Firth of Forth study area, are likely to increase. Consumption of sandeels by fish predators would increase accordingly, affecting the availability of sandeel prey for kittiwakes, perhaps to a greater extent than any impact the industrial fishery may have had in the past (Furness, 2002). Kittiwakes would appear to be the likely losers in a situation of greater competition for increasingly limited prey-fish resources. The situation would be very similar to that depicted in Figure 26, except that rather than sandeels being 'consumed' by the industrial fishery for sandeel, availability to kittiwakes would in the future be reduced through their consumption by increased populations of piscivorous fish in the area. Achieving management objectives for fisheries, fish stocks and fish communities may mean that the objectives currently proposed for seabird breeding success may need to be reassessed.

Seabird population abundance targets

A further common indicator under Descriptor 1 for seabirds, B-1, addresses 'species specific trends in the relative abundance of non-breeding and breeding marine bird species' (OSPAR, 2013b). The Criterion level target for this indicator is "*Changes in abundance of marine birds should be within individual target levels in 75% of species monitored*" (OSPAR 2013a). Current monitoring programmes provide data to estimate the abundance of 13 species of breeding seabirds in the Celtic Seas subregion and 16 species in the Greater North Sea (ICES, 2011). For each species-specific indicator of relative abundance, the supporting target is "*species-specific annual breeding abundance should be more than 80% of the baseline for species that lay one egg, or more than 70% of the baseline for species that lay more than one egg*" (OSPAR, 2013a). The setting of the baseline level is therefore of critical importance and three options have been suggested:

i. A point in the past when, based on expert judgement, anthropogenic impacts are likely to have been relatively minimal compared to the rest of the time-series; the baseline needs to reflect prevailing climatic conditions. It may prove difficult to set a baseline that meets both criteria.

ii. The mean value of the time series. This method carries the risk of a shifting baseline e.g. if a population is in long-term decline, the baseline will also decline as time goes on - so much so that the target may eventually be met, without the population recovering.

iii. Where no previous data are available: set baseline at the start of the new time-series and amend in due course - see (i) and (ii) (OSPAR, 2013a).

Setting baselines using method (i) rests entirely on interpretation of the phrase "*anthropogenic impacts are likely to have been relatively minimal compared to the rest of the time-series*". The industrial fishery for small, mainly pelagic, fish that constitute the prey of seabirds commenced in the Greater North Sea in the 1970s (ICES 2002; 2004). Sandeels *Ammodytes marinus*, the principle prey of many seabirds (Wanless *et al.*, 1998; Furness and Tasker, 2000; Daunt *et al.*, 2008), were also the mainstay of the industrial fisheries (Gislason and Kirkegaard, 1998), raising concern over the possible impact of industrial fisheries on seabird populations (Monaghan, 1992; Furness, 2002; 2003). On this basis, a baseline set sometime in the 1980s, before industrial fisheries could have impacted small pelagic fish resources too seriously, could be argued. However fisheries for human consumption had been operating in the Greater North Sea long before the industrial fishery commenced (Thurstan *et al.*, 2010). By the turn of the 20th century, annual landings of fish from the North Sea already exceeded 1 million tonnes each year and by the 1970s approximately 2.5 to 3 million tonnes of fish were being landed each year for human consumption (Daan *et al.*, 1990). Removals of such quantities of larger-sized fish would already have affected food web structure and dynamics by the 1970s, reducing the biomass of piscivorous fish in the Greater North Sea (Jennings and Blanchard, 2004). Through a trophic cascade, this could in turn have caused an increase in small pelagic fish biomass (Worm and Myers, 2003; Ward and Myers, 2005; Frank *et al.*, 2005; Frank *et al.*, 2007; Myers *et al.*, 2007; Heithaus *et al.*, 2008); which in a typical 'fishing down the food web' phenomenon (Pauly et al; 1998; Pauly *et al.*, 2000), could actually have encouraged the start of the industrial fishery.

Identifying appropriate baselines to act as the basis for setting targets for seabird population abundance indicators is clearly problematic. Even if the late 1970s/early 1980s is considered as a possible reference period for establishing baselines, it is not possible to determine whether seabird food resources at this time would be inflated through reduced predation from fish predators, a consequence of the human consumption fisheries, or depleted as a consequence of exploitation by the industrial fisheries. Given the expectation of an increase in the abundance of piscivorous fish if MSFD Descriptor 3 targets are achieved for commercial fish stocks, adoption of inappropriate baselines for setting targets for seabird abundance indicators poses a high risk of these targets failing to be met.

Similar issues arise for each of the other two methods for target setting. Each essentially results in a baseline that is derived from relatively recent seabird abundance estimates. Method (ii) would presumably be employed when monitoring time series are deemed too short to include a suitable historical reference period. Adopting the mean value of such a time series would therefore produce a baseline value for target setting that reflects the situation in recent times. Method (iii) assumes a new monitoring programme, and by using the first recorded abundance estimate as the baseline explicitly produces a target that reflects not just the recent, but also the current situation. In both cases, if meeting D3 targets for commercial fish stocks results in increased competition from fish predators for the small pelagic fish resource, so that seabird reproductive productivity declines, as discussed in the section above, then these targets may prove difficult to attain.

Seabird bycatch targets

Indicators of bycatch were included in the list of OSPAR "common indicators" evaluated by WGBIODIV (Annex 1) for both the seabird and mammal ecosystem components. Reducing, if not eliminating discarding, is a key objective for fisheries management, and WGBIODIV presumes that this chimes with the rationale underpinning these two indicators, i.e. that bird and mammal bycatch levels should be reduced. Levels of fishing mortality have declined markedly since their peak in the mid-1980s (Greenstreet *et al.*, 2011). Likewise levels of fishing activity have been considerably reduced since the turn of the century as measures specifically designed to limit fishing effort have been introduced (Greenstreet *et al.*, 2009). It is conceivable that the measures necessary to achieve key targets for D3 and targets set for fish community related D1 indicators for the Fish and Cephalopod ecosystem component may already have largely been introduced, requiring little further reduction in fishing activity. Under these circumstances, given the relationship $C = qP$ relating catch (C) (or in this case bycatch) to population abundance (P), where q, the catchability coefficient, is a constant, should seabird population abundance and distributions remain constant, then in all likelihood, bycatch levels would also remain unchanged.

Incompatibility between fisheries management D3 and marine mammal D1 targets

Seal pup production targets

Many of the same considerations discussed above in relation to seabird breeding success are also pertinent to Common indicator M-5 'harbour seal and grey seal pup production'. Again the preferred approach to target setting would involve the use of historical data to provide a baseline level representative of an acceptable situation. However, the lack of appropriate data is recognised and furthermore, even it were available, its suitability for target setting is also questioned given environmental changes (e.g. coastal developments, erosion, etc.) that may have occurred during the intervening period. Instead, a type of trends based target has been proposed: "*No statistically significant long-term average decline of ≥10% at each Management Unit*". This trends based target is unusual in that rather than aspiring to positive (recovery) trends, it sets out to avoid negative (declining) trends, and as such implicitly assumes that this indicator already meets GES; the management objective is therefore to maintain GES and avoid any deterioration in status.

Small pelagic fish, such as sandeels and sprats *Sprattus sprattus*, also constitute a major component of the diets of both grey *Halichoerus grypus* and harbour seals *Phoca vitulina* (Pierce *et al.*, 1991a; Pierce *et al.*, 1991b; Thompson *et al.*, 1991; Hammond *et al.*, 1994; Tollit and Thompson, 1996; Tollit *et al.*, 1997; Brown *et al.*, 2001). Decades of excessively high fishing mortality have reduced the biomass of the main piscivorous species globally (Christensen *et al.*, 2003; Myers and Worm, 2003) and this is particularly apparent in regions such as the North Sea (Hislop; 1996; Cook *et al.*, 1997; Jennings and Blanchard, 2004; Heath, 2005). Targets for Descriptor 3 (D3), relating to commercially targeted species, require fishing mortality rates to be reduced to levels consistent with maximum sustainable yield (F_{msy}), which should promote recovery of the biomass of these predatory fish species. In the North Sea, for example, small pelagic fish constitute a major fraction of the fish prey of many of the most depleted piscivorous fish species (Hislop *et al.*, 1991; Greenstreet *et al.*, 1998; Temming *et al.*, 2004). Increase in the biomass of predatory piscivorous fish would therefore increase predation loadings on these pelagic fish prey, and this in turn might reduce their availability to other marine top predators competing for such prey, such as grey and harbour seals (Furness, 2002). Variation in food availability, through its impact on adult female body condition, can affect pregnancy and lactation rates, and so affect annual pup production (Guinet *et al.*, 1998). In years when their preferred small pelagic fish prey were less abundant, body condition of female harbour seals in the Moray Firth, NE Scotland, was lower and pup production rates declined (Thompson *et al.*, 1996). If meeting Descriptor 3 targets for commercial fish stocks results in increased competition for the small pelagic fish resource, or in a reduction in the abundance of this resource, then this could have detrimental consequences for seal production rates and make it difficult to achieve targets of no decline.

Seal abundance targets

Common indicator M-2 addresses the 'abundance of harbour and grey seals at haul-outs and within breeding colonies'. The baseline and target setting procedure mirrors that for pup production. Recognising that data to support appropriate historic level baselines are unlikely to be available, the proposed target is "*Maintain populations in a healthy state, with no decrease in population size with regard to the baseline (beyond natural variability) and restore* *populations, where deteriorated due to anthropogenic influences, to a healthy state*". In the absence of monitoring data to support a historical baseline, some other form of evidence will be necessary to demonstrate a situation of deterioration due to anthropogenic influences. With this proviso, the general presumption again appears to be that seal abundance at haul-outs and breeding colonies is currently at GES, so the only requirement will be to ensure no decrease in population size. However, if, as discussed above, achieving fisheries management objectives for commercial fish species results in increased competition for a potentially diminished small pelagic fish resource, such that pup production declines, then it follows that population size at haul-outs and breeding colonies might also subsequently decrease.

D3 Fisheries management targets

Thus far this discussion very much reflects the arguments made above regarding incompatibility between D3 fisheries management targets and targets set for seabird indicators under D1 and D4. However, grey and common seals differ from seabirds in one very important respect – if the abundance of their preferred pelagic fish declines they can switch to alternative prey species, such as gadoids and flatfish. In years when clupeid abundance was low, harbour seals in the Beauly Firth, NE Scotland, switched to a predominantly gadoid diet (>60%, mainly whiting *Merlangius merlangus*) (Thompson *et al.*, 1996). Even if in the northern North Sea, small pelagic fish such as herring *Clupea harengus*, sprats and sandeels dominate (generally >90%) the diet of harbour seals (Thompson *et al.*, 1991; Tollit *et al.*, 1998), in waters to the north and west of the UK, and in the southern North Sea, gadoids and flatfish constitute a higher proportion of the diet (des Clers and Prime, 1996; Hall *et al.*, 1998; Brown and Pierce, 1998; Pierce and Santos 2003), testifying to the fact that harbour seals are quite capable of consuming such prey. Around 30 to 50% of the gadoid prey consumed by harbour seals exceeds a length of 30cm (Hall *et al.*, 1998; Brown and Pierce, 1998), the same size range targeted by fishermen. But even if only smaller fish were consumed, increased consumption by seal predators might still have a negative impact on gadoid recruitment. In some studies the diet of grey seals, particularly at certain times of year, has been dominated by gadoid and flat fish species (Prime and Hammond, 1990; Hammond *et al.*, 1994), and they are certainly capable of catching and consuming the largest size of fish of each species (Hammond *et al.*, 1994).

The fact that seals can switch to consuming the very species that are the object of fisheries management measures suggests that any incompatibility between targets set for seals under Descriptor 1 and commercial fish species under Descriptor 3 need not necessarily result in failure to achieve the D1 targets. Targets for seal pup production and population abundance could well be achieved at the expense of the fisheries management objectives. Off the Scottish west coast, the long-term decline in the cod stock was almost certainly attributable to over-exploitation by the fishery. Subsequent introduction of a cod recovery plan then resulted in a marked reduction in fishing mortality. However, since the 1970s, the grey seal population has increased substantially and has now levelled off at around 30,000 to 40,000 individuals. These consume around 7000t of cod each year compared with an annual landing of a few hundred tonnes. In the decade up to 2005, fishing pressure on the stock has reduced considerably, but it has been replaced by natural mortality from seal predation. This level of natural mortality is now such that the cod stock is unlikely to recover even at current minimal levels of exploitation (Cook *et al.*, 2015). A similar situation has been reported elsewhere, notably on the Scotian shelf in the north-western Atlantic. Here decades of overfishing led to a collapse in the cod stock, which in 1993, resulted in the complete closure of the fishery. Since the 1970s, grey seal populations have increased 100-fold, from 3000 to 300 000. Since each seal consumes between one and two tonnes of fish per year, this increase in seal abundance represents a substantial increase in natural mortality, and this has inhibited any recovery in the cod stock even in the complete absence of any fishing (Mohn and Bowen, 1996; O'Boyle and Sinclair, 2012; Hammill *et al.*, 2014).

Currently, the primary management objective for commercial fish stocks under Descriptor 3 is to reduce fishing mortality rates to a level compatible with long term maximum sustainable yield, with the expectation that this will bring about a substantial increase in spawning stock biomass. If indeed current population abundance and pup production rates for grey and common seals are too high, or if under current management plans for these marine mammals, their populations should increase, then it is certainly conceivable that hoped for increases in the abundance of the major fish stocks could become jeopardised.

Cetacean abundance

Common indicator M-4 addresses the "abundance at the relevant temporal scale of cetacean species regularly present". The indicator relates to four of the most common shelf-water species, minke whale *Balaenoptera acutorostrata*, harbour porpoise *Phocoena phocoena*, bottlenose dolphin *Tursiops truncatus* and white-beaked dolphin *Lagenorhynchus albirostris*, acknowledging that it is not feasible to monitor all cetaceans, particularly the more occasional visitors to European waters, and that the four listed species already have management objectives established under OSPAR, ASCOBANS, the CFP and the Habitats Directive (OSPAR, 2013a). Like the two seal species, the use of historical data to establish baseline abundance levels is considered the most optimal basis for target setting. However, it is recognised that such data are generally lacking; the historical abundance of many cetacean species is largely unknown (Roman and Palumbi, 2003) and cannot realistically be restored (where it is known to have declined) as today's marine environment is very different. Climatic changes may have had important consequences. A modern baseline has to be utilised for the species considered, such as that provided through the SCANS/CODA surveys. Therefore the proposed target is again "*Maintain populations in a healthy state, with no decrease in population size with regard to the baseline (beyond natural variability) and restore populations, where deteriorated due to anthropogenic influences, to a healthy state*".

All four cetacean species have piscivorous diets. Harbour porpoise consume a wide range of different fish, although at any one location they tend to prey on between two and four main species. Historically, porpoise diets included a high proportion of clupeids, but with the decline in the herring stock, harbour porpoises switched to a preying on whiting and sandeels (Rae, 1965; 1973). In Scottish waters, for example, where whiting and sandeels are their key prey in more recent times, harbour porpoise may consume a greater biomass of whiting than is actually landed by the fisheries. However, 99% of the whiting consumed are smaller than the legal landing size, so any impact is likely to be on recruitment to the whiting stock, rather than a direct competitive interaction with the fisheries (Santos and Pierce, 2003). In the western Atlantic, capelin *Mallotus villosus* are still a key prey species, suggesting a continuing interest in pelagic fish prey (Fontain *et* *al.*, 1994). Bottlenose dolphins in Scottish waters have a diet consisting predominantly (>84% by weight) of gadoid prey, particularly cod, saithe *Pollachius virens* and whiting (Santos *et al.*, 2001), whereas in the southern Bay of Biscay blue whiting *Micromestius poutassou*, with modal lengths of 25cm and 35cm respectively, are the main prey (Santos *et al.*, 2007). Whitebeaked dolphin also consume a variety of different fish prey. In the southern North Sea whiting and cod made up 98% by weight of their diet (Jansen *et al.*, 2010), whereas in the northern North Sea, haddock and whiting were the predominant prey (Canning *et al.*, 2008). Minke whale are also primarily piscivorous in their diets, but they tend to consume mainly pelagic species such as herring, sprats and sandeels (Pierce *et al.*, 2004; Tjelmeland and Linstrøm, 2005).

Following exactly the same logic as applied above for seabirds and seals; targets to maintain current cetacean population levels could be incompatible with proposed fisheries management measures under Descriptor 3. If successful, these measures should bring about a substantial increase in spawning stock biomass of the major commercial species, which in turn could result increased competition for the fish prey resource from piscivorous species such as saithe, cod and whiting. Conversely, since many commercial species constitute the major prey of these cetacean species, maintaining current populations, if such targets based on contempory population levels are in fact too high, could result in failure to achieve the desired increase in the biomass of commercial fish species. In essence, proposed seabird and marine mammal targets are to maintain the status quo, to ensure no declines in population size, whereas fisheries management objectives would bring about a substantial increase in the population abundance of piscivorous fish predators. These disparate sets of the targets can only be achieved if, at the present time, the fish prey resource is not limiting. If in fact predation demands on the fish prey resource are currently in equilibrium with the supply capacity of the resource, then the fish prey resource will quickly become limiting, and targets for one or more of these groups could be compromised.

Incompatibility between fish D1/D4 and marine mammal and seabird D1/D4 target

Size based metrics have been selected as amongst the most effective indicators of the impact of fishing on the state of demersal fish communities (Greenstreet, 2008), leading to the development of the Large Fish Indicator (LFI) in the North Sea (Greenstreet *et al.*, 2011), Celtic Sea (Shephard *et al.*, 2011), and southern Bay of Biscay (Modica *et al.*, 2014). Fish are non-deterministic in their growth, so within populations, increased fishing mortality rates, which shorten life expectancy and hence shift age composition towards younger fish, have a direct effect on the mean size of individuals (Beverton and Holt, 1957). But fishing is also size selective; larger species tend to be preferentially targeted and within populations of targeted species, older, larger fish are also preferentially selected, providing further impetus in the trend towards smaller size within both populations and the community as a whole (Shephard *et al.*, 2012). In all three studies, LFI trends showed initial declines, the anticipated impact of high fishing mortality on community size structure, but then subsequently followed positive trends in more recent years, indicative of recovery following remedial fisheries management to curb mortality rates.

Aquatic food webs are strongly size structured (Kerr and Dickie, 2001; Jennings and Mackinson, 2003) such that the body size of predators exceeds the size of their prey

(Greenstreet *et al.*, 1998; Scharf *et al.*, 2000; Jennings *et al.*, 2002; Jennings and Warr, 2003; Floeter and Temming, 2005). Fishing induced reductions in the size composition of fish communities therefore tend to be linked to changes in the trophic structure, implying a trend towards lower mean trophic level associated with reduced piscivore abundance (Pauly *et al.*, 2000; Jennings *et al.*, 2001). Reductions in the prevalence of piscivorous fish in fish communities around the world has been a characteristic consequence of overfishing (Christensen *et al.*, 2003; Myers and Worm, 2003; Jennings and Blanchard, 2004; Heath, 2005). Because of this link between size composition and trophic structure within fish assemblages, the LFI has been proposed as both a direct biodiversity (D1) and an inferential food web (D4) indicator.

The recent recoveries in the LFI observed in all three marine regions, which have followed reductions in fishing pressure on the demersal fish assemblages, are therefore indicative of changes in assemblage trophic structure; this implies an increase once again in the abundance of piscivorous fish within the three demersal fish communities. In the Greater North Sea, this increase in the biomass of demersal piscivorous fish inferred from the LFI recovery has also been demonstrated directly (Figure 28). Despite these recent recoveries, in all three areas where an LFI has been defined, the targets have still to be met. In all three areas, analysis of the relationship between the LFI and variation in fishing mortality suggests that the requisite management measures have already been put in place so further increase in the LFIs are anticipated and it is expected that LFI targets will be met within the next 5 to 10 years.

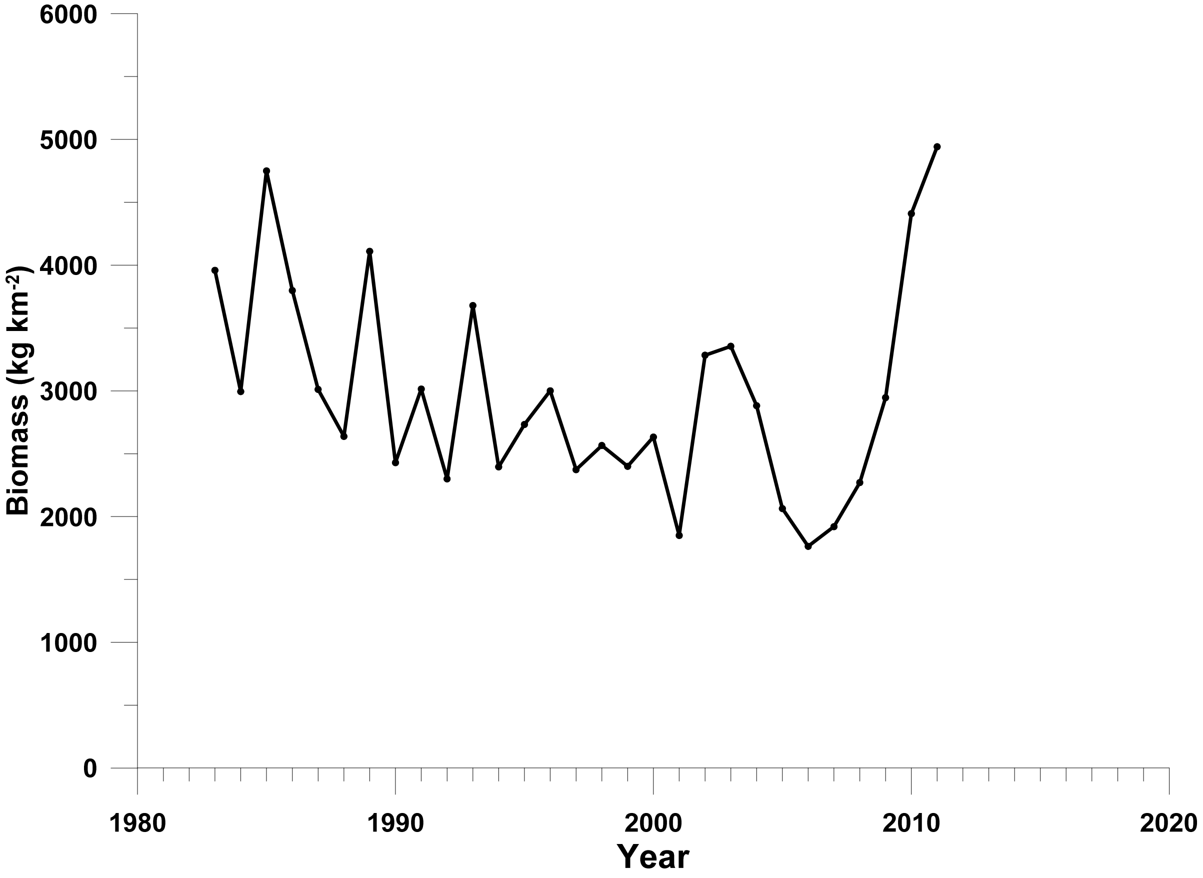


Figure 28. Trends in the biomass of demersal piscivorous fish in the Greater North Sea derived from first quarter International Bottom Trawl Survey data corrected to take account of variation species- and length-related catchability in the GOV trawl. Assignment to the demersal piscivore guild takes account of species- and length-specific variation in the diet.

In this respect, targets for fish community indicators supporting Descriptors 1 and 4 are entirely consistent with management objectives for the commercials stocks under Descriptor 3; indeed the management measures implemented to achieve the Descriptor 3 targets and exactly the same measures necessary to achieve the Descriptor 1 and 4 fish community targets. Consequently, following same logic espoused above, successful management to recover the LFI and to meet targets set for this indicator in each marine region, management that would raise the biomass of piscivorous fish, is likely to increase competition for the fish prey resource and make it more difficult to meet current GES targets for seabird and marine mammal top predators. Cascading effects down the trophic levels of marine food webs have been described (Frank *et al.*, 2007; Heithaus *et al.*, 2008; Baum and Worm, 2009) and the consequences of changes in the biomass of piscivorous fish that arise as a result of remedial management put in place to meet MSFD commitments are likely to have a range of unexpected consequences that might affect capacity to meet some targets.

Meeting D3 indicator targets might go some way to achieving the majority of fish community related targets for D1 Fish and Cephalopod indicators, but it may cause problems with regard to meeting targets for other ecosystem components (Dickey-Collas *et al.*, 2014). For example, targets may need to be revised for indicators related to population abundance, reproductive productivity, distribution range and distribution pattern within the range for both seabirds and marine mammals (Daunt *et al.*, 2008; Sharples *et al.*, 2009).

Incompatibility between fisheries management D3 and benthic invertebrate D1 and D6 targets

Two common indicators have been proposed, BH-1 "typical species composition" and BH-2 multi-metric indices"; both provide a measure of change in the species composition, structure and function of benthic invertebrate communities. In both instances, current prevailing conditions are deemed to represent a disturbed state impacted by fishing activity (Reiss *et al.*, 2009; Reiss *et al.*, 2010b). A more historic state is therefore considered more appropriate as the baseline for setting targets OSPAR 2013a).

North Sea plaice

In the southern North Sea, major fisheries have targeted flatfish species that are important predators of benthic invertebrate species. Haddock, one of the principal roundfish species targeted in the northern North Sea is also an important predator of benthic invertebrates (Greenstreet *et al.*, 1997; Heath, 2005; Schückel *et al.*, 2010). A recent compilation of spawning stock biomass data for plaice suggests that plaice biomass in the North Sea has reached a level not seen in 90 years (Figure 29). Plaice constitute approximately 50% of the total biomass of demersal benthivorous fish in the North Sea, and the recent marked increase in plaice biomass is also reflected in a similar increase in the combined biomass of all demersal benthivorous fish in the region (Figure 30). Estimates of demersal benthivorous fish biomass are currently higher than observed at any other time in the Q1 IBTS time series, by a factor of two on average. If the trend displayed by plaice alone, is also indicative of the overall demersal benthivore trend, then this could imply that predation pressure imposed by demersal benthivorous fish in the benthic invertebrate communities may also be higher than any at any point in the last 90y. The recent increase in plaice biomass is directly linked to the marked reduction in fishing mortality that has occurred over the last 10 years (Figure 31). Recent reductions in fishing activity (Greenstreet *et al.*, 2009) are also likely to have benefitted many of the other less common non-targeted benthivorous species as well (Piet *et al.*, 2009). Any reduction in the levels of mortality experienced by benthic invertebrates as a direct consequence of contact with fishing gears that might be anticipated through meeting D3 targets and reducing fishing activity could simply be countered by an increase in rates of natural mortality as the population abundance of their fish predators increases (Frid *et al.*, 1999). D1 benthic invertebrate species abundance targets for Benthic Habitats indicators might therefore not be achievable, or meeting them could be substantially delayed.

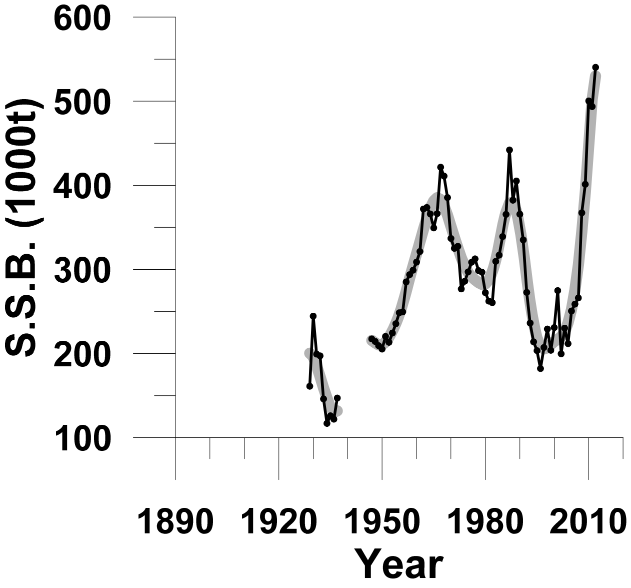


Figure 29. Long term trends in the spawning stock biomass of plaice in the Greater North Sea. Black line and dots shows actual estimates and grey line show Loess smoother fitted to the data.

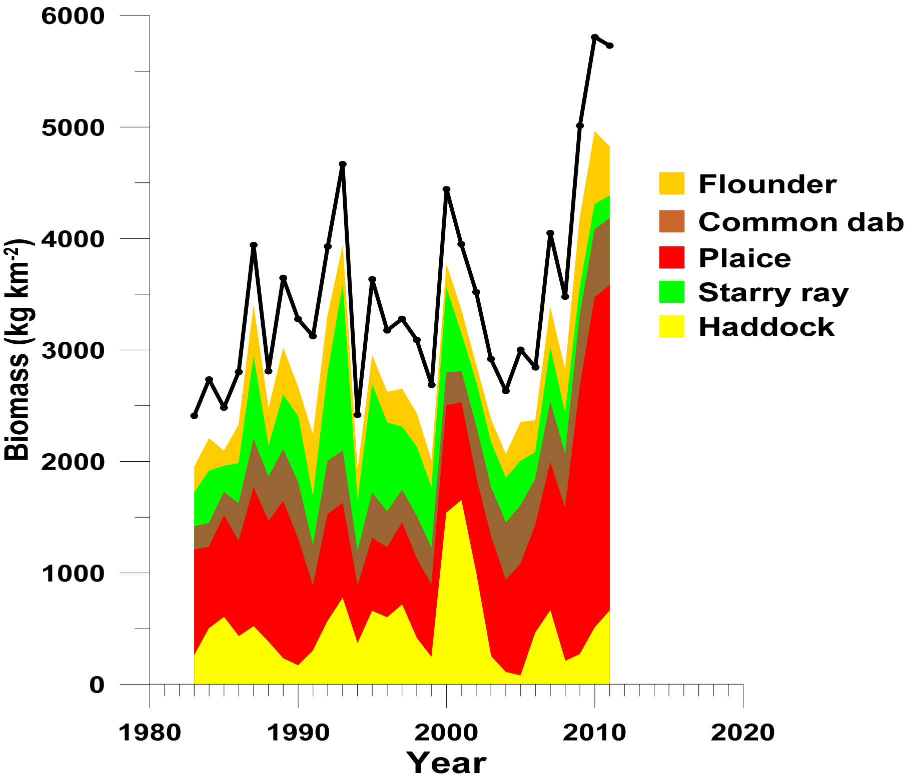


Figure 30. Trends in the biomass of demersal benthivorous fish in the Greater North Sea derived from first quarter International Bottom Trawl Survey data corrected to take account of variation species- and length-related catchability in the GOV trawl. Assignment to the demersal benthivore guild takes account of species- and length-specific variation in the diet. Contributions to the total guild biomass (black line and dots) by each of the five most abundant species is indicated.

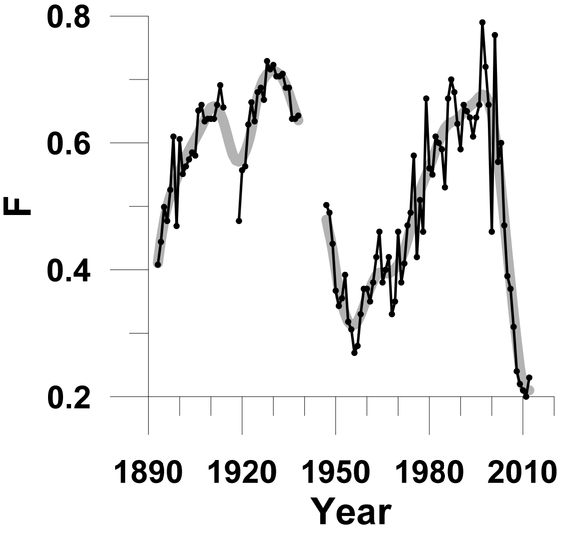


Figure 31. Long term trends in fishing mortality on plaice in the Greater North Sea. Black line and dots shows actual estimates and grey line show Loess smoother fitted to the data.

Concluding comments

In this section we have identified several situations where, because of differences in the baseline or reference periods being applied, targets set for different indicators may not be compatible with one another. Because of these inconsistencies in the target setting rationale, achieving targets for a set of indicators for one ecosystem component may make achieving current targets for indicators related to another ecosystem component extremely difficult. When undertaking an IEA to support implementation of the MSFD, if targets are not met, rather than simply assuming that management has been inadequate and simply introducing additional measures that are even more stringent and restrictive of human activities, the first reaction should perhaps be to question whether the targets set were appropriate in the first place.

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