ICES WGECO REPORT 2010

ICES Advisory Committee

ICES CM 2010/ACOM:23

Report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO)

7–14 April 2010

Copenhagen, Denmark

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. 2010. Report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO), 7–14 April 2010, Copenhagen, Denmark. ICES CM 2010/ACOM:23. 225 pp.

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Executive summary

A major focus for WGECO in 2010 was helping to build a sound scientific basis for implementation of the Marine Strategy Framework Directive (MSFD). Some Terms of Reference drafted in 2009, prior to progress reports from the ICES-JRC Task Groups being available, did not mention the MSFD explicitly. However as work of the Task Groups was completed, the close parallels between those WGECO Terms of Reference and needs for science support arising from the Task Group and Management Group Reports became clear. In those cases, the work of WGECO was correspondingly expanded to address aspects of the MSFD that were relative to the topics of the ToR (particularly ToRs a) and b)). Consequently, as a package, the contents of ToRs a) on integrated ecosystem assessments and selection of indicators (Section 3), b) on recovery and reference points for sustainability (Section 4), and d) on combining information across indicators in assessing status (Section 6), provide much of the guidance required for experts to conduct assessments of Good Environmental Status (GES) that simultaneously are ecologically *appropriate* for the ecosystems in the regions where they are done, and ecologically *consistent* across regions even when different indicators and/or reference levels were used.

Relative to the work in Section 3, for implementing the marine strategy framework directive, no integrated ecosystem assessment (IEA) is formally required. However, the necessary assessments are expected to include an explicit description of the relationships between pressure and state, multiple impacts and socio-economic aspects. These assessments should also be able to provide the basis for developing marine strategies including programmes of measures. Assessments sufficient to meet these needs will have to have many of the characteristics reviewed for sound integrated ecosystem assessments. The guidance for conducting sound IEAs, particularly the detailed guidance on how to choose indicators, will be invaluable guidance for doing these tasks in support of the MSFD. Moreover, by following a common approach as developed in Section 3, the possibility for consistency and comparability of assessments across marine regions is increased greatly. WGECO developed a framework to address the development of IEAs in a consistent manner.

This Report also considers how the diversity of marine ecosystems, uses, socio-economic settings and availability of data across marine regions, means that rigid methodological guidance on setting assessment benchmarks cannot be expected to be an appropriate strategy to achieve consistency among assessments. Rather, the consistency is achieved by the *functional equivalence* of the elements assessed, indicators chosen (Section 3) and reference levels established (Section 4). Section 3 explains how ecosystem elements and indicators can be considered functionally equivalent when they are appropriate for measuring status of a pressure, structural or functional property or process that is of *similar ecological significance* across ecosystems, even if the exact indicators or properties differ across ecosystems. Section 4 explains how reference levels can be considered functionally equivalent if they reflect the same level of sustainability, or risk of serious harm, across ecosystems, even if the *value* of the indicator that reflects this level varies across ecosystems.

In Section 6 the previous elements are brought together into a framework through which Member States can assess "good environmental status" (GES). It lays out six necessary steps (i) how to evaluate the list of ecosystem components required, (ii) how to evaluate the list of pressures and drivers required, (iii) how to identify the key interactions between ecosystem components and pressures, (iv) how to select indicators for those key interactions identified in (iii), (v) how to set reference points for these indicators, and (vi) how to combine information across indicators at various

levels of integration. It also includes two additional pieces of technical guidance. The first is an approach for setting ecologically consistent reference levels for pristine conditions, in the few Descriptors that the MSFD implies should not be impacted as opposed to being used sustainably. The second is for step (vi), with guidance both on analytical/technical aspects of integration of information across indicators, and on aspects of the scientific *processes* necessary and appropriate for such integration.

In considering relevance of indicators, we undertook a preliminary analysis of the match of the candidate list of indicators suggested by the COM Elements of a Decision, against important aspects of ecosystem structure, function and process. The preliminary table produced has revealed some interesting trends in terms of the utility of some of the candidate indicators, which if reviewed in time should help inform the process of indicator selection by Member States (before July 2010).

In Section 7 (ToR e)), the focus was on large scale "integrated ecosystem management plans (IEMP)" that are in use or are in an advanced stage of development. The focus was on the Norwegian national Barents Sea Ecosystem-based Management Plan (in force since 2006) and the HELCOM Baltic Sea Action Plan.

The differences between the two plans examined in detail in Section 7 highlight that the concept of an IEMP covers a very wide range of types of plans. They can differ in the level of the objectives set, in the degree to which they contain specific management provisions, and whether the provisions are oriented more at outcomes or at regulatory actions. This diversity is neither a strength nor a weakness of the IEMPs; just an inescapable consequence of the extended social and governance processes that are central to development of the plans. The Barents Sea Plan was a product of a single country; the Baltic Plan was the product of several countries coordinated through a formal regional seas organization. These governance and social differences are rooted in cultures, national laws and regional agreements and are not likely to converge soon. Therefore it is appropriate to plan for a continued diversity of contents in the category of IEMPs.

This inescapable diversity in the contents of IEMPs makes it unrealistic and probably unhelpful to pursue a line of evaluation that would suggest that there is some single "right" level of science input to IEMPs, or even some single "right" degree of linkage between the plans and the science available for their development. However, our review revealed other pathways to explore and provides constructive guidance for the relationship between science and the development of the IEMPs. These pathways build on some of the positive conclusions that also came from our consideration of the IEMPs. WGECO proposes to continue this line of work in 2011.

There are many examples of methods to assess threat or risk of impact of particular activities, including well established risk assessment frameworks. In most cases, however, these are for either single or multiple pressures, on one type of marine component, or for single pressures on multiple components. Where they cover multiple pressure/component interactions, the assessments are usually done independently for each pressure/component interaction resulting in a potential lack of consistency between them. There have been other attempts to develop integrated approaches, e.g., REGNS, and the Australian 3-tier ecosystem risk assessment framework, as well as research programs undertaking comparative evaluations of threats to ecosystems, e.g., the IndiSeas project. The aim in the OSPAR QSR assessment was to try to simultaneously assess the importance of different pressures across multiple components in a number of very different marine ecosystems. The process was designed to use coherent definitions and, particularly, thresholds between classes of response (i.e., good, moderate or poor) to provide consistency between the ecosystem areas and components. In Section 8 (ToR f)), WGECO reviewed the methodology used by the OSPAR workshop on the development of Chapter 11 of the QSR 2010 (Utrecht workshop) and considered the improvements that could be made to the thresholds between different assessment classes, including any scientific basis for proposed thresholds. WGECO concluded that if an integrated assessment such as the OSPAR QSR was to be the science basis for implementation of the MSFD, it would need to include:

- An explicit description of the relationships between pressure and state;
- A common approach that will ensure consistency and comparability across marine regions;
- Include multiple impacts and socio-economic aspects (e.g. in the context of the MSFD);
- Include consideration of data uncertainty or knowledge gaps.

Further, it would need to follow the process outlined briefly in Section 8, and more extensively in Sections 3, 4 and 6. This can and should be done within the domain of the MFSD, and the results of the process should significantly improve any future QSR approach. WGECO concluded that the OSPAR QSR assessment methodology is applicable at all the spatial scales mentioned in the request.

As an element of this request, WGECO were asked to consider whether it is possible to extend this methodology to support the assessment of plankton communities (ToR f)) and considered that inclusion of the plankton community would clearly enhance the holistic and integrative nature of the OSPAR assessment (see Sections 3 and 8 of this report), but would require changes to some aspects of the methodology.

WGECO continued to work on the proportion of large fish indicator (LFI; ToR c)) and results are presented in Section 5. The LFI time-series for the demersal fish community of the North Sea was updated to 2008. The LFI has continued to rise and now stands at 0.22 against an EcoQO limit of 0.30. The relationship between the LFI and fishing mortality (F) averaged across the commercial species was examined. Changes in F in one year produced the expected response in the LFI, not in the first January following, but in the second. Long-term trends in the LFI and F were related, but with an asynchrony of between 12y and 18y reflecting the integration of all the processes initiated following any change in F. Theoretical process-based partial ecosystem models intended to provide a sound scientific basis for management advice are still under development. However, preliminary simulations using one such model suggest that it may simply be sufficient to fish the main commercial species at F_{PA} to achieve the EcoQO for the North Sea demersal fish community by 2020.

It has always been intended that the LFI be used in other marine regions; the EcoQO approach developed for the North Sea was a pilot study with the expectation by OSPAR that it would rolled out to other OSPAR marine regions. To this end an analysis of spatial variation in that LFI in different sub-regions of the North Sea was undertaken. It was hoped that, by understanding the processes underlying spatial variation in a data-rich region where the demersal fish community has been long studied, the lessons learnt would aid this “roll-out” process. Finally, initial analyses were performed in three “case-studies” applying the LFI in different marine regions. These studies, and the North Sea sub-regional study, illustrate the need to “tune” the metric to enhance its signal-to-noise ratio in different areas, where both the fish communities and the pressures and ecological drivers controlling community composition and structure might differ markedly. The need to develop a formal process to do this was therefore clearly highlighted, and the lessons learnt in the North Sea pilot study should be invaluable to this end. WGECO proposes to focus on this aspect of their work in 2011.

WGECO also responded to a request to review the environmental interactions of wave and tidal energy generation devices (marine wet renewables, Section 9). Tidal barrages in locations where they will generate significant levels of power will alter tidal processes over large areas (potentially regional sea scales) although there is scope for mitigation of many of the direct ecological impacts. Many of the sites suitable for use will be RAMSAR sites. While turbine life may be of the order of 2 decades the barrage structure will potentially have a design life of >100 years. Tidal barrages represent a major modification to the coastal environment impinging on natural processes, including bird feeding areas and the migration routes of catadromous and anadromous fish, and many maritime sectors. These changes need to be balanced against the potential to deliver very significant quantities of low carbon energy. The scale of the construction projects for barrages and fences is potentially large and many of the major impacts associated with this phase, for example noise from pile driving, can be mitigated by careful planning, for example by avoiding critical times of year for marine mammals.

Tidal stream devices to generate significant power output will occupy large areas of sea for several decades. Although devices are likely to be well spaced within a farm, the sites themselves will have a large spatial footprint. Adoption of effective marine plans by Member States and within Regional Seas will be necessary to address this concern. Wave energy collectors have the potential to alter water column and sea bed habitats and by changes in the wave environment cause changes some distance from the installation. The scale of the impacts is limited and will scale with the size of development and vary depending on the nature of the location selected. Effective marine spatial planning and rigorous licensing requirements will do much to minimise the possible environmental impacts. Most effects would be reversible, fairly rapidly, if an installation was removed. Tidal stream devices and wave energy collectors themselves will have generally only local impacts, similar to those already encountered during routine marine construction activities. Potential concerns with impacts to pelagic organisms still need to be resolved, but are not considered a serious threat at this stage. The fact that wave energy and tidal stream devices are still in the experimental/trail phases means that there is no data on the environmental effects of commercial developments. Appropriate scientific studies should therefore accompany the licensing of the first commercial scale installations.

1 Opening of the meeting

The Working Group on Ecosystem Effects of Fishing Activities (WGECO) met at ICES HQ, Copenhagen, from 10.00 Wednesday 7 April–17.00 Wednesday 14 April 2010. The list of participants and contact details are given in Annex 1.

1.1 Acknowledgements

WGECO would particularly like to thank Claus Hagebro and Helle Gjeding Jørgensen of the ICES Secretariat for their support in enabling the meeting to run smoothly.

2 Adoption of the Agenda

The meeting Agenda (Annex 2) was adopted on April 7th and the meeting proceeded according to the workplan presented in the first two plenary sessions by the Subgroup Leaders. Throughout the meeting, subgroup meetings were scheduled to allow for member participation in a number of subgroups to the degree possible. Daily updates were provided by the Subgroup Leaders in plenary session and as text was finalized it was presented in plenary. Therefore, all of the content of this report pertaining to the Terms of References was fully reviewed in plenary sessions of the WGECO.

3 ToR a) Assess the development of integrated ecosystem assessments

a) Assess the development of integrated ecosystem assessments, in particular focusing on how assessments will be used for the MSFD and considering the use of the IOC's (in press) best practice recommendations. This assessment would include a gap analysis in terms of the availability of suitable state and pressure indicators.

3.1 Introduction

An ecosystem approach to management should provide a comprehensive framework for marine resource and activities decision making. Integrated ecosystem assessments (IEAs) are a critical element to support an ecosystem-based marine strategy. According to IOC's 'Assessment of Assessments' (AoA, UNEP and IOC-UNESCO 2009) an assessment consists of "*formal efforts to assemble selected knowledge with a view toward making it publicly available in a form intended to be useful for decision making*"; an integrated assessment takes account of interactions and cumulative effects across pressures, activities, ecosystem components, environmental, social and economic aspects (but see WGECO definitions in Section 3.2 below).

The Marine Strategy Framework Directive (MSFD) sets up a comprehensive list of ecological descriptors and characteristics, pressures and impacts that are to be used i) to assess the environmental status of European marine waters, and ii) to elaborate marine strategies, including programmes of measures to achieve Good Environmental Status (GES) in those waters by 2020. For both purposes these descriptors, pressures and impacts need to be integrated into one or several types of IEAs. Here, we review existing IEA frameworks and provide guidance as to how IEAs may be developed to serve the MSFD.

The European Union context requires a sufficient degree of "*consistency … between marine regions or subregions of the extent to which good environmental status is being achieved*." (Paragraph 25, MSFD). Several factors contribute to the difficulty in meeting this requirement. Around the European waters, the ecosystems themselves differ intrinsically in their physics, chemistry, bathymetry, and biodiversity. The histories of uses of these ecosystems, as well as the types and intensities of present uses differ. The types of data from historical and present monitoring programs differ greatly, as does the history of marine scientific research that can provide the foundation for knowledge-based assessments of environmental status.

In trying to consistently evaluate GES across regions these differences have several important implications:

- The same list of indicators is inappropriate around all European seas;
- It is unrealistic to expect that some weighted combination of values on multiple indicators can produce a "number" for environmental status that has a consistent meaning around European waters;
- It unreasonable to expect any specific quantitative benchmark for a single indicator to be appropriate in all the places where that indicator may be used; even an indicator like "species richness" that can be measured for any biotic community, will vary widely with features of the habitats and oceanography, even within regions and national waters.

For these reasons, WGECO cannot provide guidance to a specific indicator suite and assessment method appropriate for the diversity of marine areas. This also implies that a gap analysis of the availability of state and pressure indicators cannot be carried out at a general level. Rather, a common framework for developing these methods and indicator suites has to be developed with the objective of ensuring consistency. Here, we go on to define the various types of assessments (Section 3.2) and consider the types of assessment required for implementing the MSFD (Section 3.3). This is followed by a review of some existing IEAs (Section 3.4) and finally, we take the best aspects of those existing methodologies and extend them into a process suited to the requirements of the MSFD (Sections 3.5 and 3.6).

3.2 Definitions of assessments

In 2007 WGECO provided the following definitions of assessment types (ICES, 2007-Section 5.1):

*"**Assessment** – the most general term. A pressure, a state or a response can be assessed, alone or in many combinations. An assessment can be the evaluation of status, or status and trends, and can be with or without reference points against which to assess status.*

***Multispecies assessment** – An assessment that includes more than one species, and includes dynamic predator-prey interactions among the species. Status and trends of the species are generally kept separate in the dynamics. It includes only biotic interactions.*

***Ecosystem Assessment** – An assessment that includes at least two trophic levels and often more than two species may be aggregated, and there are usually, but not necessarily more than one species/aggregate, in at least some trophic levels. Assessments that are called "ecosystem assessments" may or may not include abiotic influences on some or all of the biotic components being assessed. There is no established term to differentiate "ecosystem assessments" that do include effects of abiotic forcers from ones that do not. It would be useful to have such a term. Until such a term becomes established, it is important to always differentiate whether an ecosystem assessment being reviewed or reported did or did not include abiotic forcing.*

***Integrated Ecosystem Assessment** – An integrated ecosystem assessment has all major trophic levels represented and linked, although the level of aggregation of species at each level can be high or low, and may differ among levels. Integrated Ecosystem Assessments must have major abiotic forcers included dynamically. The hydrographic model may be part of the analytical tool used for the integrated ecosystem assessment, or may be run separately from the biological one, and provide drivers for a dynamic biological model. In addition to State attributes of ecosystems, Integrated Ecosystem Assessments should either estimate directly or produce outputs adequate to estimate the status and trends of the dominant Pressures and Impacts as well.*

***Integrated Ecosystem Assessments with socio-economic aspects** – This long and awkward label is used to refer to Integrated Ecosystem Assessments where it is intended that Drivers and Responses will be part of the assessment, and often when Pressures and Impacts are of equal or more interest than the bio-physical state variables. It would be useful to have established terms to differentiate integrated ecosystem assessments that primarily focus on the biological and physical components of the ecosystem from integrated ecosystem assessments that give substantial emphasis to the human dimensions of uses of the ecosystem. Such terms do not yet exist.*

*Following from the last point, there is a second partition that can be made to at least ecosystem assessments and integrated ecosystem assessments. Assessments may be relative to a single industry sector to inform policy and management of fisheries, aquaculture, marine transport, etc. **Strategic Environmental Assessments** systematically and comprehensively assess the environmental effects of a plan or programme (or policy). The objective of the SEA process is to ensure that environmental considerations are taken formally and fully into decision-*

making. In each case the sector specific assessment is intended to evaluate how well a specific industry sector can be supported by the ecosystem, and/or the size and nature of the footprint of the industry sector on the ecosystem. Consequently, for different sectors, different ecosystem components may be included, or at least disaggregated and assessed with as much accuracy and precision as possible. The differences reflect the parts of the ecosystem with which each industry sector interacts most directly. Within each sector these are likely to each be considered an "integrated ecosystem assessment", but the components, dynamics, and results may be different.

***Integrated Ecosystem Assessments for Integrated Management of all human activities in the sea**. These necessarily require Integrated Ecosystem Assessments with socio-economic considerations. They are intended to support policy and management to permit simultaneous achievement of the social and economic objectives of all industry sectors active in an ecosystem and to evaluate the total footprint of all the human activities in the sea, including cumulative effects and interactions. As such they may still give some emphasis to some ecosystem components on which particular industries depend directly or impact severely. However, they also must give emphasis to the ecosystem components and interactions (biotic or abiotic) most likely to regulate ecosystem structure and function. This can make their results differ from the results of a sector-specific "integrated ecosystem assessment" [sometimes to the surprise and consternation of the industry sector].*

Both sector-specific and multi-sector integrated ecosystem assessments have valuable uses, and again there is no established terminology for differentiating among them. It would be useful to establish such terminology, as well as terminology to clarify whether an integrated ecosystem assessment is intended to produce estimates of P and R indicators directly, or simply support their estimation outside the assessment. Without [clear language on][common terminology for use with] these issues, it can be expected that confusion and sometimes misunderstandings about what will and will not be done in and result from different assessments will be increasingly frequent."

3.3 Types of assessment(s) needed for implementing the Marine Strategy Framework Directive

The MSFD requires:

1) An initial assessment of the current environmental status of its regional seas and the impact of human activities thereon;

2) Determination of good environmental status (GES) and establishment of environmental targets and associated indicators, based on this initial assessment;

3) Implementation of a monitoring programme for ongoing assessment of the environmental status;

4) Development and implementation of a programme of measures designed to achieve or maintain good environmental status; and

5) Assessment of progress towards good environmental status and review of the effectiveness of management measures implemented in Step (4).

The initial assessment by Member States (Step 1 above) should include "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, and an economic and social analysis of their use and of the cost of degradation of the marine environment. They may use assessments already carried out in the context of regional sea conventions as a basis for their analyses" (Paragraph 24, MSFD). This suggests an ecosystem status assessment, where the ecosystem includes humans and their uses of the environment. Because this initial assessment will form the basis to identify appropriate indicators, a programme of measures, and the monitoring required to track the success of these measures in reaching GES, it needs to identify the links between environmental status and the pressures that are having an impact on it. It also needs to take account of the key interactions between ecosystem components. In this sense the initial assessment would fit under the category of "Integrated Ecosystem Assessments for Integrated Management of all human activities in the sea" as described in Section 3.2 above.

The ongoing assessment should inform Member States and the Commission of progress towards GES, and is therefore both a status and trend assessment, which also describes the changing status of marine ecosystems. As this ongoing assessment is to be used for regular updating of targets and management measures, it should include both pressure and state indicators and the linkage between them. Further, it should make explicit the effectiveness of management measures in achieving the desired changes in pressures and the subsequent changes in state. Thus the IEA approach undertaken in the MSFD initial assessment will require further extension to include the full cycle as described in Steps 1–5 above.

Finally, the Member States are to be provided with "criteria and methodological standards as to ensure consistency and to allow comparison between marine regions of the extent to which GES is being achieved". Given that it is not possible to use the same indicators, identical algorithms for suites of similar indicators, or constant quantitative benchmarks for indicators that may be widely appropriate, how can consistency be achieved? There are high-level ecosystem concepts like "integrity" and "ecosystem structure and function" that WGECO have discussed at many past meetings. These high level qualities are clearly central to GES (Paragraph 4) "'environmental status' means 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, including those resulting from human activities inside or outside the area concerned". We consider that the following steps are required to ensure consistency within any integrated ecosystem assessment methodology that would meet the requirements of the MSFD:

1) An evaluation of the components of each regional ecosystem with regard to its "*structure, function and processes*", taking account of "*natural physiographic, geographic, biological, geological and climatic factors*" which identifies the parts of that particular ecosystem that are most crucial to its ecological integrity, structure, and function. In selecting these, indicators that relate to integrated aspects of the ecosystem (e.g., those that represent food web structure) should also be considered in order to capture the interactions of components within the regional ecosystem being assessed.
2) An evaluation of the major human activities that are likely to result in pressures in each regional ecosystem (including physical, acoustic, chemical and biological pressures), which identifies the pressures likely to be causing the greatest perturbations within that ecosystem, and the scales on which those pressures are operating. Here we include the pressures associated with climate change since there is unequivocal evidence that humans are contributing to climate change.
3) Use of a scientifically peer reviewed framework (see ICES, 2006) that consists of a cross-tabulation of pressure – ecosystem component interactions that reflects which types of ecosystem components are likely to be most

impacted, or otherwise be most sensitive to the pressures identified in 2, and the pressures most likely to impact detrimentally the ecosystem components identified in 1. This cross-tabulation must also link back to the potential sources of pressures (e.g., the activity-pressure relationships identified in 2).

4) For the components and pressures that are evaluated to be most important, ensure that one or more robust and sensitive indicators are selected. Give particular attention to the interactions between the more important components from 1 and the more severe pressures from 2, which come out of the consideration in 3.

5) For each indicator, use a strategy that is appropriate to the indicators and the available data to choose a reference level, which, *for that system*, reflects:

- For state indicators, the value of the indicator at a time when pressures affecting the indicator were considered sustainable;
- For pressure indicators, the value of the indicator from a time when the ecosystem components most sensitive to the pressure were considered to be in an unimpaired state;
- If data are insufficient for the first two alternatives, the value of either type of indicator when scientifically sound analyses of historical data suggests that there is low likelihood that the structure, function or process represented by the indicator was impaired;
- If data are insufficient for the first three alternatives, the value of either type of indicator, at which theoretical or generic modelling results suggests that there is low likelihood that the structure, function or process represented by the indicator would be impaired.

The consistency is, therefore, achieved by the *functional equivalence* of the indicators and reference levels. Indicators can be considered functionally equivalent when they are appropriate for measuring status of a pressure, structural or functional property or process that is of *similar ecological significance* across ecosystems, even if the exact indicators or properties differ across ecosystems. Reference levels can be considered functionally equivalent if they reflect the same level of sustainability, or risk of serious harm, across ecosystems, even if the *value* of the indicator that reflects this level varies across ecosystems (see discussion in Sections 4 and 6 of this Report).

3.4 Integrated ecosystem assessments: existing frameworks

Several approaches to IEA have been developed with the ultimate purpose of guiding management actions. Here we summarize the work done by OSPAR and REGNS and include examples from the United States and Canada.

3.4.1 OSPAR approach

The Robinson *et al*. (2009) methodology was applied to an expert-judgement assessment of nine broad ecosystem components across the five OSPAR Regions at a workshop held in Utrecht in February 2009. Essentially, it was a qualitative assessment of the status of a number of broad ecosystem components taking into account the degree of impact of any relevant pressures on them, and using the best available data and knowledge to guide the assessment. Geo-referenced data on the distribution of state and pressure variables was provided where available and other source materials included reports and peer-reviewed papers. Where necessary, the best available information was the collective knowledge of those experts present and a confidence assessment was used to qualify this. The methodology was based on the conceptual risk-based approach of Robinson *et al*. (2008) but was modified to meet the requirements of the OSPAR Quality Status Reporting on the ecosystem status of the OSPAR regions. This meant that the assessment of resilience and resistance within the risk-based approach was considered against two reference levels instead of the original one, and that the baseline used was pre-industrial conditions (as specified in the OSPAR guidance). The reference levels (thresholds in Robinson *et al*. (2009)) were based on the (modified) Habitat's Directive Criteria for Favourable Conservation Status for Habitats and Species. They were used to set thresholds between Good and Moderate and Moderate and Poor status, and to assess the degree of impact of any relevant pressures (those that an ecosystem component was exposed to) as High, Moderate or Low.

The assessment covered most biological aspects of the ecosystem grouped into broad categories (e.g., fish, marine mammals, deep sea habitats, seabirds), but missed other components such as the plankton, marine reptiles and jellyfish. It assessed the effects of pressures on the components, but it did not explicitly assess interactions between components, nor the effects of environmental drivers (unless they were covered by pressures resulting from them). A description of the major results from the workshop, and the comments made on these by participants and observers, is given in OSPAR (2009). A review of the assessment using the Assessment of Assessment's (AoA) criteria is also given in ICES 2009, Section 6.

3.4.1.1 Strengths

1) The framework itself was well received by the participants of the workshop, including the use of a clear audit trail and confidence assessment, and the value of ensuring consistency across components and pressures was realised.

2) The process was successful in guiding a wide group of experts (over 60 participants from various discipline backgrounds and nationalities) to complete an assessment for large regions and multiple pressure/component interactions in a limited timeframe (5 days).

3) The process would allow the following questions to be answered:

   3.1) Which key pressures of human activities are likely to be responsible for the observed trends or patterns in the ecosystem components?

   3.2) Which human activities are likely to be producing the specific mix of pressures?

4) A review of the process using the AoA criteria suggested the framework scored highly in terms of relevance, and reasonably well in terms of legitimacy.

5) The framework is based on the list of ecosystem components and pressures listed in Annex III of the MSFD.

6) The use of a "worst-case" example should allow for any particularly vulnerable cases (e.g., species, habitat types) to be highlighted where they would not show up in the broad component category.

3.4.1.2 Weaknesses

1) It is not a truly integrated ecosystem assessment because the framework does not include:

   1.1) Socio-economic drivers;

   1.2) Interactions between ecosystem components;

   1.3) Environmental/abiotic drivers.

2) A review of the process using the AoA criteria suggested the framework scored poorly in terms of credibility, largely because:

   2.1) The level of aggregation of some ecosystem components was unsuitable;

   2.2) The thresholds used were inappropriate for some of the components and had no scientific basis.

3) The spatial scale of application did not match well to the threshold criteria for some ecosystem components.

4) The confidence in the assessment undertaken for some components in some regions was very low, and although a confidence assessment was included, there was some concern that the level of confidence would not be well conveyed in any final reporting based on such an assessment.

5) Although detailed instructions were given on the steps to follow in the assessment, there was some inconsistency of application between groups working on different ecosystem components. In particular, some groups used very different baseline conditions despite these being specified in the instructions.

6) This approach does not lead directly to management measures. This would require a further step.

7) The treatment of aggregate effects of different pressures on components was based on a score-based approach. The rationale for such an approach needs to be considered further.

8) There was not enough time allowed for the provision of data to the assessment process. Participants commented that they would have been much more confident with the results obtained had better data (where it does exist) been made available to them.

3.4.2 REGNS approach

A scientific expert group convened by ICES prepared a plan for how ICES could contribute to the development of an Integrated Ecosystem Assessment (IEA) for the North Sea, by undertaking a pilot study utilising marine monitoring data (Kenny *et al*., 2009). The North Sea ecosystem was defined on the basis of 114 state and pressure variables resolved as annual averages between 1983 and 2003 and at the spatial scale of ICES rectangles. The coverage of ecosystem components was limited to seabirds, plankton and fish and the assessment included a number of environmental drivers but only pressure variables related to one type of human activity – fishing. The variables were selected on the basis that they included data from a long unbroken time-series and broad spatial coverage at the scale of the North Sea.

3.4.2.1 Strengths

1) The method allows for the identification of spatial and temporal trends across many different indicators or variables. Based on this some broad spatial and temporal patterns were identified for the North Sea.

2) The 'shade plot' produced summarises patterns over many aspects of the ecosystem in one 2-dimensional picture (but see weaknesses below).

3) The methodology using relatedness (connectivity) between components can be used to explore the interactions of components and of the effects of environmental and human drivers on them.

3.4.2.2 Weaknesses

1) The assessment is limited to components (ecosystem and pressure) that have available time-series over similar periods and spatial scales.

2) Only the pressures (and only some of these) from fishing were included in the analysis. Using this data-driven approach it would be difficult to include a full complement of pressures and ecosystem components.

3) There is no inclusion of socio-economic data, and again, the inclusion of this would be limited by data resolution and coverage (spatial and temporal).

4) There are a number of limitations with the analyses used:

4.1) The approach is essentially correlative with all known associated drawbacks: primarily, it is difficult to interpret what is cause or effect, common consequence of a hidden factor, or what are concomitant trends just by chance;

4.2) Rodionov's 2004 sequential algorithm procedure does not allow for temporal trends in data, which would invalidate the conclusions made about regime shifts.

5) An unweighted principal components analysis gives equal weight to each variable and the distribution of variables amongst components was not equal. The 'shade plot' produced from the anomalies of the PCA eigenvalues is limited by this assumption but this is not intuitively obvious to end-users.

6) The conclusions that can be drawn from the relatedness analyses to explore interactions between components and between components and drivers are limited in scope because of the exclusion of certain aspects of the ecosystem (ecosystem components and pressures on them).

3.4.3 United States approach

In the United States (U.S.) context, an integrated ecosystem assessment (IEA) is defined as a formal synthesis and quantitative analysis of information on relevant natural and socio-economic factors, in relation to specified ecosystem management objectives (Levin *et al.*, 2009). IEAs do not necessarily supplant single-sector management; instead, they inform the management of diverse, potentially conflicting ocean-use sectors. The development of an IEA can be described as a five-stop process with a sixth step that provides monitoring feedback. These six steps are briefly described below and are linked, to the extent possible, with the steps of the MSFD (as listed above).

1. *Scoping process to identify key management objectives and constraints*. Starting from the entire ecosystem perspective the scoping step focuses the assessment on a sub-system of ecosystem components that are linked to the issues of management importance. The scoping process involves stake-holders with differing objectives, which cross ecological, social and political boundaries and who have unclear or open-access property rights on ecosystem services. The scoping process corresponds to elements of the MSFD initial assessment (Step 1).

2. *Identify appropriate indicators and management thresholds*. Indicators may track the abundance of single species, may integrate the abundance of multiple species, or serve as proxies for ecosystem attributes of interest that are less readily measured. Management thresholds can be derived from historical baseline data and/or models fit to the ecological data. Useful indicators should be directly observable and based on well-defined theory, be understandable to the general public, cost-effective to measure, supported by historical time series, sensitive and responsive to changes in ecosystem state, and responsive to the properties they are intended to measure (Rice and Rochet, 2005). The step corresponds with establishing a series of environmental targets and associated indicators in the MSFD (Step 2).

3. *Determine the risk that indicators will fall below management targets*. The goal of the risk analysis is to qualitatively or quantitatively determine the probability that an ecosystem indicator will reach or remain in an undesirable state as specified by thresholds in Step 2. Risk analysis is used to characterize the scale, intensity, and consequences of particular pressures on the state indicators, either by qualitative ranking by expert opinion or with quantitative analyses. The MSFD does not include explicitly a risk-analysis step, but a risk-based approach has recently been suggested as an appropriate aspect of prioritising management within the MSFD assessment (Cardoso *et al*., 2010).

4. *Combine risk assessments of individual indicators into a determination of overall ecosystem status*. The risk analysis quantifies the status of individual ecosystem indicators, whereas the full IEA considers the state of all indicators simultaneously. The US approach relies heavily on ecosystem models of varying degrees of complexity to provide this integration. The MSFD does not require this integrative step, or provide guidance on how to integrate multiple indicators into fewer.

5. *Evaluate the ability of different management strategies to alter ecosystem status*. Ecosystem modelling frameworks are used to evaluate the ability of different management strategies to influence the status of natural and human system indicators. Management strategy evaluation can be used as a filter to identify which measures are capable of meeting the stated management objectives. This step corresponds to an important aspect of the process of developing a programme of measures in the MSFD (Step 4).

6. *Monitoring of ecosystem indicators and management effectiveness*. Continued (and possibly enhanced) monitoring of ecosystem indicators is required to determine the extent to which management objectives are being met. A separate evaluation of management effectiveness is required to determine if management measures are having the desired effect on the pressure indicators. This step can be considered adaptive management in an ecosystem context. It corresponds to the establishment of a monitoring programme in the MSFD (Step 5).

3.4.3.1 Strengths

1) The IEA process and its objectives have been defined in published articles.

2) Provides an explicit vehicle to focus assessment and management actions across government agencies and state and federal jurisdictions.

3) Flexibility to make the management objectives and constraints specific to the region.

4) Management objectives can be determined as part of the scoping process, which allows for opportunity for increased stakeholder input.

5) IEAs can be performed at different spatial scales, ranging from Puget Sound (e.g., 100 km) to the California Current (e.g., 1000 km).
6) Includes risk assessment as an explicit step.
7) Combines risk assessments of individual indicators into a determination of overall ecosystem status. Integration is provided by ecosystem models, including pressure-state links.
8) Monitors ecosystem indicators and management effectiveness, allowing for adaptive learning.

3.4.3.2 Weaknesses

1) Lack of central guidance on the scope and core elements of an IEA (e.g., no candidate lists of state indicators and pressure indicators).
2) IEAs may become open-ended or diverted if the management objectives are not stated *a priori*.
3) Because of this the indicators and modelling framework may be inappropriate for answering the management questions.
4) Heavily dependent on ecosystem models (Ecopath, Ecosim, Atlantis), even in data-rich regions, to provide the integration of state indicators and to evaluate management measures (is the real ecosystem being assessed or the model of it?).
5) The IEA process does not provide guidance for setting reference points for ecosystem attributes; in the US reference points for fish stocks, marine mammals and endangered species are set by law in the corresponding acts.
6) The IEA process can help to justify existing monitoring programs but has no mandate to initiate additional monitoring to fill data gaps.

3.4.4 Canadian approach

The Ocean Action Plan (OAP 1; http://www.dfo-mpo.gc.ca/oceans-habitat/oceans/oap-pao/index_e.asp) developed under Canada's Oceans Act, (http://www.dfo-mpo.gc.ca/acts-loi-eng.htm) included plans to develop Integrated Management Plans for five Large Ocean Management Areas (LOMAs). The governance processes for integrated management plans was to be based on inclusive planning and consultation "tables" where multiple departments from federal, provincial and territorial, and municipal governments would all participate, along with representatives of a range of stakeholders from ocean industries, social, environmental and business organisations, academia, and communities. At these tables mixes of human activities would be discussed which would together provide the suites of social and economic benefits sought by the participants, while ensuring healthy and productive ecosystems. These consultations were to be informed by Ecosystem Assessment and Overview Reports (EOARs).

Early in the EOAR process, it was decided to take a criterion-based approach to identifying conservation priorities for each LOMA. Initially, *a priori* criteria would be set, on scientific grounds, for *ecologically and biologically significant areas* (EBSAs) (DFO, 2004), *ecologically and biologically significant species and community properties* (EBSSs) (DFO, 2006), *depleted species*, and *degraded areas*. Degraded areas were dropped part way through the process because of jurisdictional concerns. For depleted species, it was agreed that the assessments already being done by DFO relative to limit reference points, and assessments done by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC), were sufficiently rigorous and broad in coverage to be the source of candidate Conservation Objectives associated with depleted species. The criterion-based approach to assessing ecosystem status had the advantage of making the choice of Conservation Objectives transparent and objective.

The criteria were all relative ones, such that within each class (EBSA, EBSS, Depleted species) the Conservation Objectives were ranked by ecological priority. However, as work progressed, it became clear that for the ecological importance of the Conservation Objectives to be consistent within and among LOMAs, guidelines were needed on how to merge Conservation Objectives from the three separate lists (for example, how to rank a badly depleted species of fish relative to a rare habitat type, and relative to a key foraging species). This guidance, and associated guidance on how to phrase the high priority outcomes from application of the criteria as Conservations Objectives that met the criteria above, was provided by DFO (2007; 2008).

The EOARs were completed for all five LOMAs, and in most cases within the scheduled time frame (DFO-nd). Although the governance process has gone in a different direction than envisioned at the start of OAP 1, the EOARs have been used in a number of subsequent applications where some form of integrated science knowledge was needed as a basis for action, such as the Ecosystem Status and Trends Reports required for meeting commitments for reporting of biodiversity under the CBD.

3.4.4.1 Strengths

1) The criteria give an objective and documentable way to select some parts of the ecosystem for focus, whether during more in-depth assessments, prioritizing conservation initiatives, planning research, or other subsequent activities. They are relative criteria so that a series of areas or species can be ranked on the criteria, rather than providing a binary in-out decision, so the selection of areas gives more flexibility to follow-up actions.

2) The criteria can be applied by a rational science-based process, where the discussion and conclusions can follow established science peer review processes for reliability, plausibility, and balanced treatment of uncertain or contradictory evidence.

3) Application of the criteria necessarily requires “integration” of information across the ecosystem components; for example identification of “forage species” or evaluating the “fitness consequences” associates with an area.

4) The criteria that led to specific places and species being ranked highly can remain associated with the places or species in the follow-up activities, so the ecological contexts and interpretations remain associated with the assessment or management uses of the higher ranking places and species.

5) Because of #4, the results of application of the criteria can give clear direction to the nature of indicators that should be used and the properties that should be reflected in the position of the reference levels on the indicators. This removes much of the arbitrariness from selection of indicators and reference levels.

6) The criteria have been shown to be usable with a variety of qualities and quantities of data, from strictly narrative traditional knowledge to fine-scale and geo-referenced datasets.

7) The science basis for the individual criteria is well-documented, and can be revised and revised as needed, as further scientific knowledge accumulates.

8) The criteria seem to be pretty stable and robust, particularly the more widely-used spatial criteria. For the place-based criteria, a literature review of 47 different publications which considered criteria for ecologically or biologically significant areas concluded that there was very little functional difference in the criteria across publications, although different publications used different phrasing for some criteria (Deardon and Topelka, 2006).

3.4.4.2 Weaknesses and limitations

1) Because using the criteria produces a relative ranking of areas, or species, some other process has to decide at what point down the ranking it is appropriate to stop, if the decision is "include in a follow-up activity" (whether the activity is a more in-depth assessment or prioritization for enhanced conservation measures).

2) Although the *a priori* criteria give structure to discussions about which parts of an ecosystem are the most important to include in follow-up measures, they do not fully protect such discussion from selective advocacy. People with a special interest in a particular ecosystem component or pressure can still build partisan cases for (or against) their component of pressure, and try to push their objective to the top (or bottom) of the priority list through partisan evidence.

3) The criteria that have been used to date address only ecological importance and function. There are no provisions for criterion-based evaluations of pressures. There is no conceptual reason why criterion-based approaches to pressures could not be done, but to our knowledge it has not been attempted.

4) The approach defers consideration of the interactions of pressures and components to a step *after* application of the place and species criteria. It also defers consideration of social and economic aspects of decision-making to steps after the application of the place, space (and, possibly in future, pressure) criteria. The approach has no specific guidance on how social and economic factors should be considered, beyond that policy and management decisions should be risk-averse relative to places and species that are ranked highly relative to the criteria.

5) When data (or other information sources) are patchily distributed, criterion-based approaches are often biased towards identifying more high-priority areas or species in the places or ecosystem components that most information rich. This weakness is not unique to criterion-based assessment approaches, but these approaches also have it.

6) For comprehensive assessments, it will be necessary to apply at least separate criteria for places that are ecologically significant and for species and/or community properties that are ecologically significant. This produces at least two separate lists of priorities, and a set of meta-criteria or rules are needed for merging the independent of different types of features.

7) Although there are tested sets of criteria for ranking places and species, there are no equivalent criteria for ranking pressures. The concept makes sense, and the information needed to develop such criteria for pressures could be assembled, reviewed, and synthesised into pressure criteria. However the task has not been done.

3.4.5 Summary

No single IEA approach described above fulfils all the requirements of the MSFD process (Table 3.4.5.1). Although the U.S. approach covers all important components (ecosystem components, socio-economics, pressures (anthropogenic and natural) and interactions), the lack of guidance on how to set reference levels means that there could be poor consistency between and within assessments. The dependence on models to provide integration within the ecosystem requires a high degree of confidence that the models adequately describe the important ecosystem processes. The holistic nature of the MSFD requirements means that the best information available on some components will not be at the level that could drive any modelling approach. Thus even if the model ensures integration across components, some parts of it might not be well supported by data and underlying assumptions might remain largely untested. An expert-judgement type approach will be required where data are poor. However, the overall six step process described in the U.S approach (Levin *et al.*, 2009) does set out nicely the full cycle required to implement the assessment steps to fulfil the MSFD.

The REGNS approach is an entirely data-driven approach requiring large amounts of continuous time-series data. As described in Step 5 of Section 3.3.1, time-series data will be important in setting reference levels for some indicators used within an IEA, but the analyses utilised in the REGNS approach are not suitable for the general requirements of the MSFD assessments.

Elements of both the OSPAR and Canadian approaches can, however, be developed and taken forward in guiding a suitable IEA approach that would meet the MSFD assessment requirements and these are described below.

Table 3.4.5.1. Summary of the coverage of important requirements of an Integrated Ecosystem Assessment for the MSFD (as outlined in Section 3.3) by a number of existing IEA applications as described in Section 3.4.

Approach	Pressure-State Links	Consistency and Comparability	Status/Trend	Missing Components
OSPAR	Approach aims at identifying significant links	Limited (no scientific basis for ref levels)	Primarily status	Socio-economics, Environmental forcing, Interactions among components
REGNS	Exploratory (correlative)	Limited (no reference level)	Trend	Those with no time-series data
US	Functional relationships assumed in assessment model	Limited (no guidance for objectives, indicators or reference levels)	Status and trend	None
Canada	Missing	Guidance from conservation objective	Status and trend	Link with socio-economics, Pressures

3.5 The way forward - the initial assessment

Following the five steps outlined in Section 3.3 above, we describe in further detail the elements of an IEA that could be used by MSFD contracting parties to ensure a consistent and credible initial assessment of their regional ecosystems, and use this to assess GES and set monitoring and management priorities.

3.5.1 Evaluation of ecosystem components

The MSFD already provides guidance on the characteristics of the ecosystem that should be considered in the initial assessment (Table 1, Annex III). These include aspects of the physical and chemical features, habitat types and biological features of the ecosystem. It is clear from this that Member States (MSs) must at least describe the status of their regions covering these aspects, but the degree of aggregation (or disaggregation) within characteristics is not clearly expressed. For example, although characteristics of fish populations are listed, any properties of fish above the level of population are not specified and this may mean that MSs focus on populations of commercial or characteristic species, and do not consider functional groups or community level properties that take account of other species that may still be of ecological significance within the regional ecosystem. Below we provide some guidance on the selection of ecosystem features, attributes or properties that would provide representation of the characteristics specified by Annex III and ensure that the initial assessments provide information at a level that will be useful to them.

3.5.1.1 Physical and chemical features

This will be a description of the natural physiographic, geographic, geological and climatic factors in the region and should include information on any observed natural variability or cycles in these features. This information will set some important boundaries for the appropriate spatial and temporal scales to consider habitat types and biological features since their distribution will be related to the physical and chemical variables.

3.5.1.2 Habitat types

The list of Characteristics to be considered describes three different sets for Habitat types:

1) Here the seabed and water column features should be classified into units based on the prevailing physico-chemical and hydrographic regimes of the region being assessed (as determined from 3.5.1.1). Habitat classification schemes such as EUNIS will be useful here but the level of aggregation chosen (e.g., which EUNIS level) will depend in each case on: (i) the scale at which there is information on distributions of habitat types; (ii) the scale at which there is information on key pressures on habitats (see 3.5.2), and (iii) the time available to carry out the IEA. MSs must recognise that an IEA carried out at a very coarse resolution of habitat types (e.g., landscape scale) will only identify priorities for monitoring and management at that scale (see Section 8 of this report). However, by also considering 2 and 3 below, MSs should also identify any priorities at a finer habitat scale related to either protected habitats or those which are ecologically significant and at high risk of degradation due to the distribution of pressures.

2) The MSFD requires that MSs provide distributional information on any habitat types recognised as being protected under any other Community legislation (such as the Habitat's and Bird's Directives) or any international Conventions (Table 1, Annex III) found in the region or sub-region being assessed. The information on these habitat types is likely to be at a much finer resolution than for the broad assessment of representative habitats as described in 1.

3) Finally, the MSFD recognises that there may be habitats not dealt with under 2, that *"by virtue of their characteristics, location or strategic importance,*

merit a particular reference". The Canadian approach provides a method for selecting Ecologically and Biologically Significant Areas (DFO, 2004) and the CBD, 2009 reports consistently successful experiences with using a variety of similar criteria and approaches for identifying areas of high conservation priority (CBD, 2009). This experience would be useful to apply here.

3.5.1.3 Biological features

We group the list of characteristics to be considered into four different types for Biological features:

1) Information is required at the population level for fish, marine mammals, reptiles and seabirds. For the latter three categories this seems logical for the limited numbers of species that will be recorded in each region or subregion. For the fish, it will not be practical to provide information at the species level for all species representing the biodiversity in those areas (see 4). For fish, commercial species should be considered at the population level (as this information will be required later to assess GES under Descriptor 3 of the MSFD), and any species meeting the criteria under 2 and 3 below should also be considered at the population level (where possible).

2) The MSFD requires that MSs provide a summary of "*the population dynamics, natural and actual range and status of other species occurring in the marine region or sub-region*" protected under any other Community legislation (such as the Habitat's and Bird's Directives) or any international Conventions (Table 1, Annex III, MSFD).The information on these species may be at a much finer resolution than for the broad assessment of other broad species groups as described in 1. In many cases there will not be information available on some of the attributes (e.g., population dynamics) specified for the rare species that are listed.

3) The MSFD requires that MSs provide a summary of the "*temporal occurrence, natural and actual range and status of any non-indigenous, exotic or where relevant, genetically distinct forms of native species*", where present in the marine region or sub-region being assessed.

4) For the biological features not captured at the species level under 1–3 above, a broad assessment of representative features is required. This should include information on plankton (phytoplankton, zooplankton), fish assemblages, benthos, macroalgae and angiosperms. In these cases characteristics should be described using integrative indicators that pick up the ecologically significant aspects of the features such as indicators on primary production, forage species, and distributions and variability in important habitat-structuring features. Again, some useful guidance is given here in the Canadian approach where criteria are provided for selecting indicators that capture Ecologically and Biologically Significant Species and Community Properties (DFO, 2006). It will not be appropriate or necessary to provide species level information for these groups unless they meet the requirements under 1–3 above. We give further guidance in Section 3.5.4 on the indicators that would be useful in capturing broader information on key aspects of ecosystem structure, function and process.

3.5.2 Evaluation of the pressures

The MSFD provides a list of pressures that should be considered in the initial assessment (Table 2, Annex III). WGECO have much experience in matching pressures to human activities and natural drivers (ICES 2005; 2006). Member States should use the list of pressures specified by the MSFD (Table 2, Annex III) and match these against any human activities and drivers, including climate change, that occur in the region being assessed (following the process outlined in ICES) and taking account of transboundary effects. Transboundary effects include those where the drivers may occur at a coarser spatial scale (e.g., climate change) or outside of the region (e.g., source of a dispersive pollutant). It is essential that the list of pressures can be linked back to all human drivers so that monitoring and management priorities can be identified specific to the pressures that can actually be managed.

3.5.3 Use of a framework to identify key pressures and components

MSs are required to provide some broad summary information on all aspects of the ecosystem components identified following the process described in 3.5.1 (an Ecosystem Overview), but to identify GES, management and monitoring priorities, the integration of information is required. Having documented the lists of the relevant ecosystem components and pressure/driver combinations in their marine regions (or sub-regions) being assessed, MSs should use a framework to identify the key pressures on their components. It is the descriptors related to those components subject to pressures affecting their sustainability that are most likely to fail to meet GES (see Table 2.1a in Cardoso *et al*. (2010) for a match of ecosystem components (the characteristics of the MSFD) to the GES descriptors), those activities causing pressures having significant effects on components that will require management measures, and those components contributing to failures in meeting GES (and the pressures contributing to this) that will require monitoring.

WGECO have worked for several years on frameworks to identify the key pressures on ecosystem components (ICES 2005; 2006; 2007; 2008). Using the lists of the relevant ecosystem components and pressure/driver combinations in their marine regions (or sub-regions) we suggest the following steps to identify key pressures, and components at risk due to the effects of single or multiple pressures:

1) Using a matrix of all pressures against components (like the example given in Table 4.2.5.1, ICES (2005) but including the pressure/driver combinations identified in 3.5.2 and components identified in 3.5.1), consider whether there is any overlap between a pressure and an ecosystem component and mark off any where no overlap occurs. The overlap must account for both temporal and spatial factors.

2) Where overlap occurs, identify key interactions using the following criteria which have been developed from ICES (2006, Section 4.2.4):

- Spatial extent:

What is the spatial scale of overlap of the pressure and component, relative to the scale of the component (e.g., if assessing pressure *k* against the component 'habitat type *i*', where the component covers an area *j*, what is the extent of area *j* subject to pressure *k*)? Based on this, classify the extent of the pressure relative to the component as either Widespread (W) or Local (L).

- Degree of impact

Where component *i* is subject to pressure *k* what is the intensity of the impact on component *i* where they overlap? Here the best available information (from published literature where available) on the effects of different pressures on components should be consulted, and this interpreted in light of the distribution and intensity of the pressure in the region relative to the component being considered. Degree of impact should either be considered as Acute (A) or Chronic (C), where Chronic interaction should be used to describe a pressure that lasts for a long period of time or is marked by frequent recurrence, but where even the cumulative effects may not lead to any or a significant proportion of component level mortality or destruction. It may also include indirect effects to a component (e.g., changes in growth rates brought about by a change in temperature or decreased productivity of the benthos due to reduced productivity from the plankton based on increased turbidity levels). Acute impacts should be defined as relatively short but intense and instantaneous interactions that cause mortality or destruction to a component at a high proportion of the component or populations included.

- Recovery potential of components

The recovery potential of components should also be taken into account whereby those components with longer periods of recovery should be given higher priority than those with rapid periods of recovery (taking into account the spatial extent of impact). Ecosystem components with no capacity to recover are of particularly high priority. Any components that would not recover within two assessment cycles would be deemed to have a long recovery period.

The concept of recovery is discussed in detail in Section 4 of this report, and MSs should consult this where further understanding is required.

- Key interactions

Key interactions are deemed to have occurred where they are described as: (i) Widespread and Acute for any level of recovery potential of components; (ii) Local and Acute where components have long recovery periods; (iii) any pressure interactions where components are judged to have little or no capacity for recovery, or components are listed under other Community Directives or international conventions as requiring protection.

3) For any pressure where there is at least one key interaction with a component, suitable indicators will need to be selected as described in 3.5.4 below.

4) For any component where there is at least one key interaction with a pressure, suitable indicators will need to be selected as described in 3.5.4 below.

5) For any component not identified in 4, but where there are interactions with several pressures, aggregate effects must be considered (see 3.5.3.1 below). Where the aggregate effects of pressures may themselves lead to acute impacts on a component, suitable indicators will need to be selected as described in 3.5.4 below.

Previous exploration of such an approach (ICES, 2006) suggests this process will not highlight a restrictively high volume of interactions.

3.5.3.1 Aggregate effects of pressures

The risk and consequences of aggregate impacts[1] on ecosystems due to the presence of multiple pressures is discussed in a number of theoretical and practical ecological studies but still its quantification is a difficult task. In ICES 2007, some suggestions were made for methods that would allow aggregation of impacts from a particular pressure across components, or aggregation of impacts across all pressures within a single component. These were based on simple scoring approaches and a basic sensitivity analysis was presented on the effect of different levels of aggregation of scoring systems on the overall outcome.

Recent studies to explore the nature of interactions of multiple pressures show that aggregate impacts are additive (i.e., are summed) for pairs of pressures (Crain *et al.*, 2008) and that synergistic effects are generally more common than additive ones (Darling and Côté, 2008). This highlights the need to evaluate the complexity and range of uncertainty in assessing aggregate impacts of human pressures.

Assuming that multiple activities act independently within a system, recent studies by Ban and Alder, 2008 and Halpern *et al.* (2008), modelled aggregate impacts as the additive accumulation of impacts of individual activities combining a measure of ecosystem component sensitivity and the risk of occurrence of an activity. In contrast, Stelzenmüller *et al.* (2010) used generic pressure categories exerted by human activities and developed a range of models that quantify the risk of aggregate impacts on marine habitats. More precisely, their geospatial modelling framework used the footprint and intensity of a number of human pressures, measures of habitat sensitivities to those pressures and a process that allowed the alteration of the importance of single pressures. This resulted in a number of scenarios for risk of aggregate impacts with numerical results other than the addition of single pressures. This framework shows a high level of flexibility as it can be applied at any spatial scale and adapted to different pressure categories when suitable data are available. Moreover, depending on the spatial scale of its application it can be modified to focus on multiple activities rather than on multiple pressures by omitting some steps.

The aggregate impact of multiple activities was recently modelled at the scale of ecoregions and benthic habitats for Canada's Pacific area assuming a linear decay from the origin of the activities (Ban *et al.*, 2010). The authors considered specifically deep pelagic waters and shallow pelagic waters and give therefore another example on how different scenarios for the risk of aggregate impacts can be developed. Common limitations to all of the above listed studies are the lack of experimentally assessed information on the sensitivity of ecosystem components and a more comprehensive knowledge on the interactions of human activities. Especially when more information on the latter becomes available current modelling approaches can be developed further to assess the risk of aggregate impacts also on the basis of synergistic and antagonistic effects of multiple human activities.

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[1] In this report we use "aggregate impacts' to refer to the effects of several pressures at once acting on an ecosystem component, to differentiate such impacts from the common interpretation of "cumulative impact" as the impact of a single pressure acting over a long period of time.

3.5.4 Selection of indicators

Having identified the key interactions between pressures and components in regions being assessed following the steps outlined in Section 3.5.3, there is then a requirement to select indicators for those interactions that could be used to assess (1) whether GES would be met (Section 3.5.4.1 below), and (2) to prioritise monitoring and management. In terms of prioritising monitoring and management, the process outlined in Section 3.5.3 will identify the pressures and components requiring consideration. Since all of the ecosystem components are important aspects of the biodiversity of Europe's regional seas (see match of characteristics to GES Descriptor 1 (Biodiversity) in Table 2.1a of Cardoso *et al*. (2010)), this suggests that any components highlighted to be at risk (key interactions in the process described in Section 3.5.3) should be considered further in terms of selecting indictors.

In 2007 (ICES, 2007 Section 5.5), WGECO highlighted the need to select an appropriate set of indicators that could track the monitoring and management requirements for the key pressure/component interactions. In some cases indicators of pressures, activities and state variables would be required, but in others a pressure indicator alone would suffice. In addition, for each component where more than one key pressure has been highlighted, different indicators may be required. For example seals are likely to be affected quite differently by fishing activity and heavy metal contamination. For the former, population size might constitute the appropriate indicator of state, while for the latter, contaminant levels in blubber samples would better convey the changes in component state resulting from the activity.

3.5.4.1 Selection of indicators to describe GES

The process described in 3.5.3 is a way of consistently identifying the characteristics (components) of the ecosystem that may be impaired by the pressures in the system. In Cardoso *et al*. (2010) the characteristics of the ecosystem and the pressures listed in Annex III of the MSFD are matched against the GES descriptors (Tables 2.1a and b). This reveals that there is an uneven coverage of the descriptors by the pressures and characteristics that MSs are required to provide information on in their initial assessment. For some descriptors, information will only be relevant from indicators from a small number of characteristics and/or pressures. MSs must consult with Tables 2.1a and b from Cardoso *et al*. (2010) to ensure that they have coverage of enough indicators to address each of the GES descriptors. GES will ultimately be assessed against indicators of these descriptors (and the process of taking and combining this information is described in Section 6.3 of this report).

For some GES descriptors many possible indicators have been proposed by Task Groups requested to suggest indicators for this purpose (see summary in Cardoso *et al*., 2010). Since the MSFD promotes the sustainable use of the marine environment while safeguarding its processes, functions and structures (Article 1, para 8, MSFD), WGECO considered that an appropriate additional step in the indicator selection process should be a check of the coverage of the attributes of ecosystem structure, function and process against the indicators recommended by the Task Groups.

Here we aimed to match the available list of candidate indicators with important ecosystem processes, functions and structures to derive a first qualitative assessment of the indicators and clear guidance on the selection process. Expert judgement was used to undertake this assessment and we note that the full complement of expertise required to complete this with a high degree of confidence, was not available. The scoring shown in the table produced (Technical Annex Section 3.7) is to be considered "work in progress". Some parts of it need to be validated with a wider range of experts, but the Technical Annex is illustrative of the type of cross-tabulation that is envisioned. As such, the summary of the distribution of indicators across functions produced is also for illustrative purposes and should be viewed as such.

To complete the assessment the following steps were undertaken and could be repeated with a fuller complement of expertise at a later date:

1) The joint ICES and JRC task descriptor groups produced a Management Group report (Cardoso *et al*., 2010) which outlined in detail the indicators defined for each GES descriptor. This list, comprising 85 indicators (many of them actually classes of indicators), has therefore been used as the baseline for the development of the comparison. For this assessment the indicators describing hydrographical alteration have been omitted as we could not find any rationale for the indicators suggested and they all overlapped with indicators already proposed under other descriptors.

2) Commonly accepted ecosystem structures, processes and functions have been amalgamated from a number of key studies (Frid *et al*., 2008; Hussain *et al*., 2010) to one list of properties (Table 3.5.4.1.1).

3) In a third step, a matrix was produced which maps the suit of indicators against the selected ecosystem processes, functions and structures (Technical Annex Section 3.7). Expert judgment was used to evaluate if the measured property outlined by each indicator delivered on the respective process, function or structure. Thus all indicators have been distributed into three categories: Y (Yes; the indicator measure informs the ecosystem property), N (No; the indicator measure does not inform the ecosystem property), and P (Partial; more specification is required to inform the ecosystem property). The summary of the matrix allowed identification of both ecosystem properties that are captured best by the current list of indicators and the indicators that inform most ecosystem properties.

Table 3.5.4.1.1. List of ecosystem properties and related categories comprising ecosystem structure, function, and derived taken from Frid *et al*. (2008) and Hussain *et al*. (2010).

Ecosystem properties	Category: Structure (St), Function (F), Process (P)
Trophic structure	St F
Biologically mediated habitat	St F
Physical habitat	St
Resistance	St
Recoverability	St
Organism health	St
Production	St
Biological diversity	St
Water Chemistry	St
Energy cycling	P
Nutrient cycling	P
Active Transport	P
Passive transport	P
Productivity	P

The selected ecosystem properties reflecting processes, structures and functions were ranked on the basis of how well they captured the candidate indicators (Table 3.5.4.1.2). Thus from a total of 85 indicators, 32 related clearly to the physical habitat while only three indicators out of 85 showed a direct relation to organism health. This table gives a first indication on the distribution of candidate indicators across the important properties of ecosystems.

We also ranked the selected ecosystem properties on the basis of how many indicators were categorized as P-indicators for those properties. Table 3.5.4.1.3 indicates that there are many indicators that would require further specification in order to provide any useful information about the properties of the ecosystem. This is an important message. For indicators marked as P in the matrix (Section 3.7), the P categorization suggests that the candidate indicator group would require further specification in terms of either: (i) which ecosystem attribute it was applied to (e.g., a particular species or assemblage type), (ii) the scale it was applied to, or (iii) the indicator would require interpretation in light of further information to provide any information on the ecosystem structure, function or process in question. For example, from the total list of 85 indicators, 30 could potentially provide some information on energy cycling, but the link between the indicator and energy cycling is weak or indirect and would require further specification. We recommend that only those indicators categorized as a Y-indicator for at least one important property of the ecosystem be taken any further by MSs in choosing their set of indicators relevant to the components of the ecosystem. We note, however, that some of the candidate indicators are still useful in terms of their potential for addressing descriptors that are not important aspects of ecosystem structure, function or process (e.g., Descriptor 10 Litter and Descriptor 9 Contaminants in Fish). Indicators such as the "Amount and composition of litter ingested by marine animals" would be relevant here.

Table 3.5.4.1.2. Number of indicators categorized as Y which can be directly related to defined ecosystem properties (structure, function and processes).

ECOSYSTEM PROPERTIES	NUMBER OF Y-INDICATORS
Physical habitat	32
Passive transport	32
Recoverability	28
Resistance	26
Trophic structure	25
Nutrient cycling	18
Production	17
Energy cycling	15
Biologically mediated habitat	14
Biological diversity	14
Productivity	11
Active Transport	9
Water Chemistry	6
Organism health	3

Table 3.5.4.1.3. Number of indicators categorized as P which can be only related defined ecosystem properties (structure, function and processes) after more detailed specification of the indicator properties and measure.

Ecosystem properties	Number of P-indicators
Energy cycling	30
Productivity	29
Biologically mediated habitat	28
Organism health	25
Nutrient cycling	23
Recoverability	19
Physical habitat	18
Production	17
Resistance	14
Water Chemistry	10
Trophic structure	9
Biological diversity	9
Passive transport	6
Active Transport	3

The candidate indicators that were categorized as Y-indicators for at least one property of the ecosystem, were also ranked by the number of ecosystem properties for which they were assessed to be relevant (where a Y score was given) (Table 3.5.4.1.4). This allowed for a first assessment on the potential use of the indicators to service multiple aspects of the ecosystem.

Beyond further checking of the scoring within the cells of the Technical Annex (Section 3.7), the list of candidate indicators needs to be filtered by duplication in order to allow an assessment of indicator quality and the capability of the indicators to describe relevant processes, functions and structures.

Table 3.5.4.1.4. Candidate indicators which can be directly related to ecosystem properties (categorized as y-indicators) have been ranked by the number of associated ecosystem properties.

Code	Indicator	Number of ecosystem properties
6.1	Type, abundance, biomass and areal extent of relevant biogenic substrate	9
2.4	Magnitude of the impacts of non-indigenous species, in particular invasive species, on native communities, habitats and ecosystem	8
1.27	Interactions between the structural components of the ecosystem	7
5.7	Annual to multi-year changes in frequency and/or duration of blooms. Changes in balance of diatoms/flagellates/cyanobacteria	7
5.8	Annual to multi-year changes from fucoid/kelp to opportunistic green/brown algae	7
1.6	Population demography (e.g. body size or age class structure, sex ratio, fecundity rates, survival/mortality rates)	6
1.18	Habitat condition relates to the physical (structure and associated physical characteristics, including structuring species), hydrological and chemical conditions	6
5.4	Chlorophyll due to an increased nutrient availability, measured monthly or more frequent as appropriate	6
1.7	Population genetic structure	5
1.9	Inter and intra-specific relationships	5
1.13	The habitat condition relates to the physical, hydrological and chemical conditions	5
1.22	Community functional traits	5
2.1	Abundance and distribution in the wild of non-indigenous species and, in particular, invasive non indigenous species	5
2.2	Spreading of non- indigenous species including, where appropriate and feasible, maps of colonies distinguishing as a result of primary introduction and secondary spread	5
5.5	Increase of opportunistic macroalgae	5
6.8	Extent of area with spatial or temporal hypoxia	5
1.1	Species distribution range	4
1.2	Species distribution pattern	4
1.10	Habitat distributional range	4

Table 3.5.4.1.4. Continued.

Code	Indicator	Number of ecosystem properties
1.11	Habitat distributional pattern	4
1.14	Habitat Distributional range	4
1.15	Habitat distributional pattern	4
1.19	Community species composition	4
1.20	Community relative abundance	4
2.3	Ratio between non-indigenous species and native species in some well studied taxonomic groups, e.g. fish, macroalgae, molluscs	4
3.4	Biomass indices taken from independent sources	4
4.2	Ratio of macrobenthos invertebrate to demersal fish production or biomass	4
4.4	trophic relationships within the food web	4
5.1	Nutrients concentration in the water column	4
5.6	Dissolved oxygen due to increased organic composition, measured monthly or more frequent	4
1.12	Habitat extent	3
1.16	Areal extent of habitat (area covered)	3
1.17	Habitat volume	3
1.24	Landscape Habitat composition, cover and relative proportions	3
1.25	Landscape condition relates to the physical, hydrological and chemical conditions	3
3.3	Spawning Stock Biomass	3
4.1	Ratio of pelagic to demersal fish biomass and/or production	3
4.5	Indicators for large fish (by weight)	3
6.6	Proportion of number or biomass of individuals above some specified length/size	3
6.9	Ratio of oxygen/hydrogen sulphide concentration	3

Table 3.5.4.1.4. Continued.

Code	Indicator	Number of ecosystem properties
8.1	Concentration of listed substances in the marine environment	3
8.2	Biological effects on the elements of concerned ecosystems	3
8.3	Occurrence and extent of acute pollution events (e.g. slicks from oil and oil products) and impact on biota physically affected	3
1.3	Area covered by the species (for sessile/attached species)	2
1.4	Population biomass	2
1.21	Community biomass	2
1.23	Landscape Distributional range and areal extent	2
1.26	Composition and relative proportions of ecosystem components (habitats and species)	2
3.1	Fishing mortality	2
3.5	Proportion of fish larger than a given length, e.g. the length at which 100% of the females are mature	2
3.6	Mean maximum length across all species found in research vessel surveys	2
3.7	95% percentile of the fish length distribution observed in research vessel surveys	2
3.8	Any other indicator reflecting numerically the relative abundance of old, large fish	2
4.3	Performance of key predator species using their production per unit biomass (productivity)	2
4.6	Indicators of abundance trends	2
5.2	Deviate from normal proportion of nutrient ratios (Si:N:P)	2
5.3	Water transparency due to increase in suspended algae	2
6.2	Extent of the seabed affected by human activities	2
6.4	Presence of particularly sensitive or tolerant species	2
6.7	Parameters (slope and intercept) of the size spectrum of the aggregate size composition data	2
6.10	Presence of benthic communities associated with low oxygen conditions	2

Table 3.5.4.1.4. Continued.

Code	Indicator	Number of ecosystem properties
10.1	Amount of litter washed ashore and/or deposited on coastlines	2
10.2	Amount of litter deposited on the sea-floor, in particular in shallow areas (<40m),	2
10.3	Amount of micro-particles (in particular micro-plastic) found in the water column and their potential toxicity	2
11.1	Distribution in time and place of loud, low and mid frequency impulsive sounds	2
1.5	Population abundance	1
6.3	Diversity and richness indices, based on species number and relative abundance in the benthic community	1
6.5	Use of indexes assessing functionality of the benthic system, such as such as the proportion of opportunistic to sensitive species	1

3.5.4.2 Selection of a consistent suites of indicators

In addition to being relevant to the MSFD and GES, the indicator suites selected by Member States should as far as possible meet the general criteria expected from any indicator suite in terms of quality (Rice and Rochet, 2005). We recall here the main steps of this framework for selecting a suite of indicators:

- *Determine user needs*: here this applies to the management objectives specified within the MSFD and turned into targets within each marine region.
- *Develop a list of candidate indicators*: The earlier text of this Section deals with this step.
- *Determine screening criteria*: Rice and Rochet, 2005 provide a list of nine criteria and their relative importance for different groups of users. All nine criteria are relevant to MSFD implementation.
- *Score indicators against criteria*: scoring exercises have proven to be largely subjective (Rochet and Rice, 2005); moreover, the scoring process seems to perform best with an intermediate level of detail in indicator description and the definition of criteria (Piet *et al.*, 2008). Given that the list of candidate indicators for each relevant pressure × component cell might not be very long, a simple but largely inclusive scoring process with high level criteria and a limited number of scores (3 to 5) should be appropriate.
- *Summarize scoring results*: again this should not be a very complex task within each pressure × component cell as a limited number of candidate indicators should be available. Simple graphical methods like pie-graphs should be sufficient (see Section 6.3.3).
- *Decide how many indicators are needed*: ideally just one indicator should be selected within each pressure × component cell unless separate pressure and state indicators are required for monitoring and management purposes (although the causative link between these must be known). It will rarely be necessary to have more than one indicator per cell, and in many cases one indicator will service several or many cells in the matrix.
- *Make final selection*: whereas the previous steps referred to qualities of individual indicators, this is the step where the consistency of the complete suite needs to be considered. Indicators that, isolated or combined, respond differently to different pressures should be selected to make the most sensible meaning of the available information (see Section 6.3.3.2.2). This should be helped by a conceptual model highlighting the links between all indicators of the suite (Rochet and Trenkel, 2009). This could, for example, consist of a bubble diagram with components and pressures in bubbles and arrows showing the interactions.
- *Report on the suite of indicators*: this is dealt with in Section 6.3.3 of this report.

3.6 Conclusions

According to the IOC's 'Assessment of Assessments' (AoA, UNEP and IOC-UNESCO 2009) an assessment consists in formal efforts to assemble selected knowledge with a view toward making it publicly available in a form intended to be useful for decision making; an integrated (ecosystem) assessment takes account of interactions and cumulative effects across pressures, activities, ecosystem components, environmental, social and economic aspects, or all of them (see definitions in Section 3.2).

The Marine Strategy Framework Directive sets up a comprehensive list of ecological descriptors and characteristics, pressures and impacts that are to be used i) to assess the environmental status of European marine waters, and ii) to elaborate marine strategies, including programmes of measures to achieve Good Environmental Status (GES) in those waters by 2020. IEAs can contribute to both of these purposes for integrating descriptors, pressures and impacts (Section 3.3). Here, we have reviewed existing IEA frameworks (Section 3.4) and provided guidance as to how IEAs may be developed to serve the MSFD (Section 3.5). In doing so, we have defined 5 steps to ensure that a sufficient degree of consistency can be achieved by Member States within and across marine regions and sub-regions (Section 3.3).

In the review of existing IEAs we found that none could fulfil all the requirements of the MSFD process, but the overall six step structure outlined by the U.S. approach would be invaluable in defining the full cycle required to implement the MSFD. Elements of the methodology used in both the Canadian approach and the OSPAR approach were taken forward in Section 3.5 as we felt these could contribute to better defining the assessment steps required to ensure a consistent and credible IEA process for Member States enacting the MSFD.

The issues we focused on were (i) how to evaluate the list of ecosystem components required, (ii) how to evaluate the list of pressures and drivers required, (iii) how to identify the key interactions between ecosystem components and pressures, and (iv) how to select indicators for those key interactions identified in (iii). In the indicator selection step we focused on providing guidance on how to select indicators *relevant* to the requirements of the MSFD. A clear process is described through which Member States could achieve steps (i)–(iv).

In considering relevance of indicators, we undertook a preliminary analysis of the match of the candidate list of indicators suggested by the COM Elements of a Decision, against important aspects of ecosystem structure, function and process. The rationale here was that the MSFD explicitly states the requirement to consider structure, function and process in assessing Good Environmental Status and it is important that Member States consider this in selecting their indicator sets. This preliminary analysis has revealed some interesting trends in terms of the utility of some of the candidate indicators, and we recommend that the required revisions of the table produced (Section 3.7) are undertaken in time to inform the process of indicator selection by Member States (before July 2010).

Having undertaken the steps described in Section 3.5 of this Report, there is further guidance required on how to assess GES using the indicators selected. This is covered in Section 6 of this Report.

3.7 Technical Annex: Qualitative evaluation of indicator performance with respect to ecosystem properties

			St F	St F	St	St	St	St	St	P	St	St	P	P	P	P
			Trophic structure	Biologically mediated Habitat	Physical habitat	Resistance	Recoverability	Organism health	Production	Productivity	Biological diversity	Water Chemistry	Energy cycling	Nutrient cycling	Active Transport	Passive transport
DESCRIPTOR	CODE	INDICATOR														
Biological Diversity	1.1	Species distribution range	N	P	P	Y	Y	N	P	P	N	N	P	P	Y	Y
	1.2	Species distribution pattern	N	P	P	Y	Y	N	N	N	N	N	N	N	Y	Y
	1.3	Area covered by the species (for sessile/attached species)	N	P	P	Y	Y	N	P	N	N	N	P	P	N	N
	1.4	Population biomass	P	P	P	P	P	P	Y	P	N	N	Y	N	N	N
	1.5	Population abundance	P	P	P	P	Y	P	P	P	N	N	P	P	N	N
	1.6	Population demography (e.g. body size or age class structure, sex ratio, fecundity rates, survival/mortality rates)	P	P	P	Y	Y	Y	Y	Y	N	N	Y	P	N	N
	1.7	Population genetic structure	N	N	N	Y	Y	P	N	N	Y	N	N	N	Y	Y
	1.8	Population health	?	?	?	?	?	?	?	?	?	?	?	?	?	?
	1.9	Inter and intra-specific relationships	Y	N	N	P	P	N	Y	P	Y	N	Y	Y	N	N
	1.10	Habitat distributional range	N	P	Y	Y	Y	N	P	P	N	N	P	P	N	Y
	1.11	Habitat distributional pattern	N	P	Y	Y	Y	N	N	N	N	N	N	N	N	Y
	1.12	Habitat extent	N	P	Y	Y	Y	P	P	N	N	N	P	P	N	N
	1.13	The habitat condition relates to the physical, hydrological and chemical conditions	N	P	Y	Y	Y	P	N	P	P	P	P	Y	N	Y
	1.14	Habitat Distributional range	N	P	Y	Y	Y	N	P	P	N	N	P	P	N	Y
	1.15	Habitat distributional pattern	N	P	Y	Y	Y	N	N	N	N	N	N	N	N	Y

			St F	St F	St	St	St	St	St	P	St	St	P	P	P	P
			Trophic Structure	Biologically mediated Habitat	Physical habitat	Resistance	Recoverability	Organism health	Production	Productivity	Biological diversity	Water Chemistry	Energy cycling	Nutrient cycling	Active Transport	Passive transport
DESCRIPTOR	CODE	INDICATOR														
	1.16	Areal extent of habitat (area covered)	N	P	Y	Y	Y	P	P	N	N	N	P	P	N	N
	1.17	Habitat volume	N	P	Y	Y	Y	N	N	N	N	N	N	N	N	N
	1.18	Habitat condition relates to the physical (structure and associated physical characteristics, including structuring species), hydrological and chemical conditions	N	Y	Y	Y	Y	P	N	P	P	P	P	Y	N	Y
	1.19	Community species composition	Y	N	P	Y	Y	N	P	P	Y	N	P	P	N	N
	1.20	Community relative abundance	Y	N	P	Y	Y	N	P	P	Y	N	P	P	N	N
	1.21	Community biomass	N	N	P	P	P	N	Y	N	N	N	Y	P	N	N
	1.22	Community functional traits	P	P	P	P	P	N	N	N	Y	N	Y	Y	Y	Y
	1.23	Landscape Distributional range and areal extent	N	N	Y	N	N	N	N	N	N	N	P	P	N	Y
	1.24	Landscape Habitat composition, cover and relative proportions	N	P	Y	N	N	N	N	N	Y	N	P	P	N	Y
	1.25	Landscape condition relates to the physical, hydrological and chemical conditions	N	N	Y	N	N	N	N	N	P	P	P	Y	N	Y
	1.26	Composition and relative proportions of ecosystem components (habitats and species)	P	P	P	P	P	N	Y	N	Y	N	P	P	N	N
	1.27	Interactions between the structural components of the ecosystem	Y	P	N	Y	Y	P	Y	P	Y	N	Y	Y	N	N

			St F	St F	St	St	St	St	St	P	St	St	P	P	P	P
			Trophic Structure	Biologically mediated Habitat	Physical habitat	Resistance	Recoverability	Organism health	Production	Productivity	Biological diversity	Water Chemistry	Energy cycling	Nutrient cycling	Active Transport	Passive transport
DESCRIPTOR	CODE	INDICATOR														
Non-indigenous species	2.1	Abundance and distribution in the wild of non-indigenous species and, in particular, invasive non indigenous species	Y	P	P	Y	Y	P	P	N	P	N	P	P	Y	Y
	2.2	Spreading of non- indigenous species including, where appropriate and feasible, maps of colonies distinguishing as a result of primary introduction and secondary spread	Y	P	P	Y	Y	P	N	N	N	N	N	N	Y	Y
	2.3	Ratio between non-indigenous species and native species in some well studied taxonomic groups, e.g. fish, macroalgae, molluscs	Y	N	N	Y	Y	N	N	N	Y	N	N	N	N	N
	2.4	Magnitude of the impacts of non-indigenous species, in particular invasive species, on native communities, habitats and ecosystem	Y	P	N	Y	Y	P	Y	Y	Y	N	P	P	Y	Y
	2.5	Biopollution Level (BPL) index	N	N	N	N	N	N	N	N	N	N	N	N	N	N
commercially exploited fish and shellfish	3.1	Fishing mortality	N*	N	N	N	N	N	Y	P	N	N	Y	N	N	N
	3.2	Ratio between catch and a biomass index	N	N	N	N	N	N	N	N	N	N	N	N	N	N
	3.3	Spawning Stock Biomass	Y	N	N	N	N	P	Y	P	N	N	Y	N	N	P
	3.4	Biomass indices taken from independent sources	Y	N	N	N	N	P	Y	P	N	N	Y	N	Y	P

			ST F	ST F	ST	ST	ST	ST	ST	P	ST	ST	P	P	P	P
			TROPHIC STRUCTURE	BIOLOGICALLY MEDIATED HABITAT	PHYSICAL HABITAT	RESISTANCE	RECOVERABILITY	ORGANISM HEALTH	PRODUCTION	PRODUCTIVITY	BIOLOGICAL DIVERSITY	WATER CHEMISTRY	ENERGY CYCLING	NUTRIENT CYCLING	ACTIVE TRANSPORT	PASSIVE TRANSPORT
DESCRIPTOR	CODE	INDICATOR														
	3.5	Proportion of fish larger than a given length, e.g. the length at which 100% of the females are mature	Y	N	N	N	P	N	N	Y	N	N	N	N	N	N
	3.6	Mean maximum length across all species found in research vessel surveys	Y	N	N	P	P	N	N	Y	N	N	N	N	N	N
	3.7	95% percentile of the fish length distribution observed in research vessel surveys	Y	N	N	N	P	N	N	Y	N	N	N	N	N	N
	3.8	Any other indicator reflecting numerically the relative abundance of old, large fish	Y	N	N	N	P	N	N	Y	N	N	N	N	N	N
	3.9	Size at full sexual maturation	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Food Webs	4.1	Ratio of pelagic to demersal fish biomass and/or production	Y	N	N	N	N	N	Y	N	N	N	Y	N	N	N
	4.2	Ratio of macrobenthos invertebrate to demersal fish production or biomass	Y	N	N	N	N	N	Y	N	N	N	Y	Y	N	N
	4.3	Performance of key predator species using their production per unit biomass (productivity)	Y	N	N	N	N	N	P	Y	N	N	P	N	N	N
	4.4	Trophic relationships within the food web	Y	N	N	N	N	N	N	P	Y	N	Y	Y	N	N
	4.5	Indicators for large fish (by weight)	Y	N	N	N	P	N	Y	Y	N	N	N	N	N	N
	4.6	Indicators of abundance trends	P	N	P	N	N	N	N	N	N	N	P	P	Y	Y
Eutrophication	5.1	Nutrients concentration in the water column	N	N	Y	N	N	P	Y	N	N	P	P	Y	N	Y
	5.2	Deviate from normal proportion of nutrient ratios (Si:N:P)	N	N	Y	N	N	N	N	N	N	N	P	Y	N	N

			St F	St F	St	St	St	St	St	P	St	St	P	P	P	P
			Trophic Structure	Biologically Mediated Habitat	Physical Habitat	Resistance	Recoverability	Organism Health	Production	Productivity	Biological Diversity	Water Chemistry	Energy Cycling	Nutrient Cycling	Active Transport	Passive Transport
DESCRIPTOR	CODE	INDICATOR														
	5.3	Water transparency due to increase in suspended algae	N	N	Y	N	N	N	N	N	N	N	N	N	N	Y
	5.4	Chlorophyll due to an increased nutrient availability, measured monthly or more frequent as appropriate	Y	N	Y	N	N	P	Y	N	N	P	Y	Y	N	Y
	5.5	Increase of opportunistic macroalgae	Y	P	P	Y	Y	N	P	P	P	N	Y	P	N	Y
	5.6	Dissolved oxygen due to increased organic composition, measured monthly or more frequent	N	N	Y	N	N	P	N	P	N	Y	P	Y	N	Y
	5.7	Annual to multi-year changes in frequency and/or duration of blooms. Changes in balance of diatoms/flagellates/cyanobacteria	Y	N	Y	Y	Y	N	P	P	P	Y	P	Y	N	Y
	5.8	Annual to multi-year changes from fucoid/kelp to opportunistic green/brown algae	Y	Y	Y	Y	Y	N	P	P	P	N	P	Y	N	Y
Seafloor integrity	6.1	Type, abundance, biomass and areal extent of relevant biogenic substrate	N	Y	Y	Y	Y	Y	Y	P	N	P	Y	Y	N	Y
	6.2	Extent of the seabed affected by human activities	N	P	Y	N	P	N	N	P	N	N	N	Y	N	N
	6.3	Diversity and richness indices, based on species number and relative abundance in the benthic community	N	N	P	P	P	N	P	P	Y	N	N	N	N	N
	6.4	Presence of particularly sensitive or tolerant species	P	P	P	Y	Y	N	N	N	P	P	N	P	P	P
	6.5	Use of indexes assessing functionality of the benthic system, such as the proportion of opportunistic to sensitive species	P	P	P	P	P	N	N	N	Y	P	P	P	P	P

			St F	St F	St	St	St	St	St	P	St	St	P	P	P	P
			Trophic structure	Biologically mediated Habitat	Physical habitat	Resistance	Recoverability	Organism health	Production	Productivity	Biological diversity	Water Chemistry	Energy cycling	Nutrient cycling	Active Transport	Passive transport
DESCRIPTOR	CODE	INDICATOR														
	6.6	Proportion of number or biomass of individuals above some specified length/size	Y	N	N	N	N	N	Y	Y	N	N	N	N	N	N
	6.7	Parameters (slope and intercept) of the size spectrum of the aggregate size composition data	Y	N	N	N	N	N	N	Y	N	N	P	N	N	N
	6.8	Extent of area with spatial or temporal hypoxia	N	N	Y	N	Y	P	N	P	N	Y	P	Y	N	Y
	6.9	Ratio of oxygen/hydrogen sulphide concentration	N	N	Y	N	N	P	N	P	N	Y	P	Y	N	N
	6.10	Presence of benthic communities associated with low oxygen conditions	P	N	Y	P	P	P	N	P	P	Y	N	P	P	P
Hydrographical conditions	7.1	Extent of area affected by the alteration														
	7.2	Spatial extent of benthic habitat affected by the permanent alteration														
	7.3	Changes in benthic communities and or biomass production														
	7.4	Extent of area with spatial or temporal hypoxia														
	7.5	Presence of benthic communities associated with low oxygen conditions														
	7.6	Diversity and richness indices, based on species number and relative abundance in the benthic community														

			ST F	ST F	ST	ST	ST	ST	ST	P	ST	ST	P	P	P	P
			TROPHIC STRUCTURE	BIOLOGICALLY MEDIATED HABITAT	PHYSICAL HABITAT	RESISTANCE	RECOVERABILITY	ORGANISM HEALTH	PRODUCTION	PRODUCTIVITY	BIOLOGICAL DIVERSITY	WATER CHEMISTRY	ENERGY CYCLING	NUTRIENT CYCLING	ACTIVE TRANSPORT	PASSIVE TRANSPORT
DESCRIPTOR	CODE	INDICATOR														
	7.7	Presence of particularly sensitive or tolerant species														
	7.8	Changes in habitat functions due to altered hydrographical conditions (e.g. spawning areas, breeding and feeding areas and migration routes of fish, birds and mammals)														
Concentrations of contaminants	8.1	Concentration of listed substances in the marine environment	N	N	Y	N	N	P	N	N	N	Y	N	N	N	Y
	8.2	Biological effects on the elements of concerned ecosystems	N	N	Y	P	P	Y	P	Y	N	P	P	P	N	N
	8.3	Occurrence and extent of acute pollution events (e.g. slicks from oil and oil products) and impact on biota physically affected	N	P	Y	N	P	P	P	P	Y	P	P	P	N	Y
Contaminants in food	9.1	Actual levels that have been detected	N	N	N	N	N	P	N	N	N	N	N	N	N	N
	9.2	Number of contaminants for which exceeding levels have been detected	N	N	N	N	N	N	N	N	N	N	N	N	N	N
	9.3	Frequency where the regulatory levels are exceeded	N	N	N	N	N	N	N	N	N	N	N	N	N	N
marine litter	10.1	Amount of litter washed ashore and/or deposited on coastlines	N	P	Y	N	N	N	N	N	N	N	N	N	N	Y
	10.2	Amount of litter deposited on the sea-floor, in particular in shallow areas (<40m),	N	P	Y	N	N	N	N	N	N	N	N	N	N	Y

			ST F	ST F	ST	ST	ST	ST	ST	P	ST	ST	P	P	P	P
			TROPHIC STRUCTURE	BIOLOGICALLY MEDIATED HABITAT	PHYSICAL HABITAT	RESISTANCE	RECOVERABILITY	ORGANISM HEALTH	PRODUCTION	PRODUCTIVITY	BIOLOGICAL DIVERSITY	WATER CHEMISTRY	ENERGY CYCLING	NUTRIENT CYCLING	ACTIVE TRANSPORT	PASSIVE TRANSPORT
DESCRIPTOR	CODE	INDICATOR														
	10.3	Amount of micro-particles (in particular micro-plastic) found in the water column and their potential toxicity	N	N	Y	N	N	N	N	N	N	N	N	N	N	Y
	10.4	Amount and composition of litter ingested by marine animals	N	N	N	P	P	P	N	P	N	N	N	N	N	N
Introduction of Energy (noise)	11.1	Distribution in time and place of loud, low and mid frequency impulsive sounds	N	N	Y	P	P	P	N	P	N	N	N	N	N	Y
	11.2	Continuous low frequency sound	N	N	Y	P	P	P	N	P	N	N	N	N	N	P

3.8 References

Ban, N., Alder, J. 2008. How wild is the ocean? Assessing the intensity of anthropogenic marine activities in British Columbia, Canada. Aquatic Conservation Marine and Freshwater Ecosystems, 18: 55.85.

Ban, N.C., Hussein, M.A., Ardron, J.A. 2010. Cumulative impact mapping: Advances, relevance and limitations to marine management and conservation, using Canada's Pacific waters as a case study. Marine Policy, doi:10.1016/j.marpol.2010.01.010.

Cardoso, A.,c, Cochrane, S., Doerner H., Ferreira, J.G., Galgani, F., Hagebro, C., Hanke, G., Hoepffner, N., Keizer, P.D., Law, R., Olenon, S., Piet, G.J., Rice, J., Rogers, S.I., Swartenbroux, F, Tasker, M., and van de Bund, W. 2010. Marine Strategy Framework Directive, Management Group Report. Luxembourg Office for Official Publications of the European Communities, 59 pp.

CBD, 2009. Expert Workshop On Scientific And Technical Guidance On The Use Of Biogeographic Classification Systems And Identification Of Marine Areas Beyond National Jurisdiction In Need Of Protection. UNEP/CBD/EW-BCSandIMA/1/2.

Crain, C.M., Kroeker, K., and Halpern, B.S. 2008. Interactive and cumulative effects of multiple human stressors in marine systems. Ecology Letters, 11: 1304–1315.

Darling, E.S., Côté, I.M. 2008. Quantifying the evidence for ecological synergies. Ecology Letters 11: 1278–1286.

Dearden, P. and Topelko, K.A. 2006. Establishing Criteria for Identification of Ecologically and Boilogically Significant Areas on the High Seas. Report to Oceans Sector Department of Fisheries and Oceans, Government of Canada. available from Oceans Sector – DFO – Ottawa, Canada.

DFO. 2004. Identification of Ecologically and Biologically Significant Areas. DFO Can. Sci. Advis. Sec. Ecosystem Status Rep. 2004/006.

DFO. 2006. Identification of Ecologically Significant Species and Community Properties. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2006/041.

DFO. 2007. Guidance Document on Identifying Conservation Priorities and Phrasing Conservation Objectives for Large Ocean Management Areas. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2007/010.

DFO. 2008. Further Guidance on the Formulation, Prioritization, and Use of Conservation Objectives in an Ecosystem Approach to Integrated Management of Human Activities in Aquatic Ecosystems. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2008/029.

DFO. nd. Ecosystem Overview and Assessment reports for Eastern Beaufort Sea. Eastern Scotian Shelf, Gulf of St. Lawrence, Pacific North Coast, and Placentia bay – Grand Bank. Available on CD from DFO Oceans Sector, Ottawa, Canada.

Frid, C.L.J., Paramor, O.A.L., Brockington, S. and J. Bremner. 2008. Incorporating ecological functioning into the designation and management of marine protected areas. Hydrobiologia, 606:69–79.

Halpern B.S., Walbridge, S., Selkoe, K.A., Kappel, C.V., Micheli, F., D'Agrosa, C., Bruno, J.F., Casey, K.S., Ebert, C., Fox, H.E., Fujita, F., Heinemann, D., Lenihan, H.S., Madin, E.M.P., Perry, M.T., Selig, E.R., Spalding, M., Steneck, R., and R. Watson. 2008. A global map of human impact on marine ecosystems. Science 319: 948–952.

Hussain, S.S., Winrow-Griffin, A., Moran, D., Robinson, L.A., Fofana, A., Paramor, O.A.L., and C.L.J. Frid. 2010. An ex ante ecological economic assessment of the benefits arising from marine protected areas designation in the UK. Ecological Economics, 69:828–838.

ICES. 2005. Report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO). WGECO Report 2005 ICES Advisory Committee on Ecosystems, ACE:04, 146pp.

ICES. 2006. Report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO). WGECO Report 2006 ICES Advisory Committee on Ecosystems, ACE:05, 179pp.

ICES. 2007. Report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO). ICES Document CM 2007/ACE:04. 162pp.

ICES. 2008. Report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO). WGECO Report 2008 ICES Advisory Committee on Ecosystems, ACE:41, 269pp.

ICES. 2009. Report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO). WGECO Report 2009 ICES Advisory Committee on Ecosystems, ACE:20, 190pp.

Kenny, A.J., Skjoldal, H.R., Engelhard, G.H., Kershaw, P.J., Reid, J.B. 2009. An integrated approach for assessing the relative significance of human pressures and environmental forcing in the status of Large Marine Ecosystems. Progress in Oceanography 81: 132–148.

Levin, P.S., Fogarty, M.J., Murawski, S.A., Fluharty, D. 2009. Integrated ecosystem assessments: developing the scientific basis for ecosystem-based management of the ocean. PloS Biology 7: 23–28.

OSPAR. 2009. Report of the Workshop on Biodiversity Assessments for the QSR 2010. Meeting of the Environmental Assessment and Monitoring Group (ASMO) 20–24 April.

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 J. Mar. Sci. 65(8): 1449–1455.

Rice, J., Rochet, M.J. 2005. A framework for selecting a suite of indicators for fisheries management. ICES J. Mar. Sci. 62: 516–527.

Robinson, L.A., Rogers, S., Frid, C.L.J. 2008. A marine assessment and monitoring framework for application by UKMMAS and OSPAR – Assessment of Pressures (Contract No: F90-01-1075 for the Joint Nature Conservation Committee). University of Liverpool, Liverpool and Centre for the Environment, Fisheries and Aquaculture Science, Lowestoft. 108pp.

Robinson, L.A., Karman, C., Rogers, S., Frid, C.L.J., 2009. Methodology for assessing the status of species and habitats at the OSPAR Region scale for the OSPAR Quality Status Report 2010. (Contract No: C-08-0007-0085 for the Joint Nature Conservation Committee). University of Liverpool, Liverpool and Centre for the Environment, Fisheries and Aquaculture Science, Lowestoft. 33pp.

Rochet, M.J., and Rice, J. 2005. Do explicit criteria help selecting indicators for Ecosystem-based fisheries management? An experimental test. ICES Journal of Marine Science 62(3): 528–539.

Rochet, M.J., and Trenkel, V.M. 2009. Why and how could indicators be used in an ecosystem approach to fisheries management? In: Future of Fishery Science in North America. Beamish R.J., Rothschild B. (eds). Springer. pp. 209–226.

Rodionov, S.N. 2004. A sequential algorthm for testing climate regime shifts. Geophysical Research Letters 41. 4pp.

Stelzenmüller, V., Lee, J., South, A., and Rogers, S.I. 2010. Quantifying cumulative impacts of human pressure on the marine environment: ageospatial framework. Marine Ecology Progress Series 398: 19–32.

UNEP and IOC-UNESCO. 2009. An assessment of assessments, findings of the group of experts. Start-up phase of regular process for global reporting and assessment of the state of the marine environment including socio-economic aspects. ISBN 978-92-807-2976-4, United

Nations Environment Programme and the Intergovernmental Oceanogrphic Commission of UNESCO, Valetta, Malta.

4 ToR b) Data analyses required to examine the relationships between perturbation and recovery capacity

b) Address the data analyses required to examine the relationships between perturbation and recovery capacity, or some other element of "cost", for ecosystem components where there is still little information available (e.g., habitats, benthos, fish assemblages, communities in general);

There is not one, 'high level' definition for the term recovery in relation to marine ecosystems and their components. Moreover, among different sources, the terms 'recovery', 'rebuilding' and 'restoring/restoration' may be applied synonymously, or differentiated in a variety of ways not consistent among sources (ICES, 2009). In addition, the terms can be applied to diverse ecosystem components, from single fish stocks to suites of similar species, to communities and whole ecosystems. The basis for a consistent interpretation of any of the terms when applied to individual populations, to community components or whole communities, or to ecosystems has not been established. Last year WGECO reviewed the use of these terms in a variety of international agreements and concluded that "*the 'recovery' definition is more broadly usable in an operational context. WGECO further notes that in much of its fishery advice, ICES has included the phrase that recovery should be expected to be "rapid and secure". ("Rapid" is applied taking account of the normal dynamics of the properties being monitored. "Rapid" for herring is not the same as "rapid" for beluga). WGECO agrees that the condition of "rapid and secure" is an important aspect of recovery*" (ICES 2009). It also discussed the benchmarks for concluding that recovery has been achieved, particularly in the context of resilience of ecosystem properties, and concluded "*WGECO consider that provided the majority of the functions are present at a level similar to before disturbance, and that the key species are present, a system may be considered to have recovered.*"

Since that meeting, however, there have been two further major developments that may affect our conclusions about the interpretation of "recovery" in policy and management. One is the major ICES/PICES/UNCOVER Symposium on Rebuilding Depleted Fish Stocks –Biology, Ecology, Social Science and Management Strategies in November 2009. Although the papers from that Symposium are not yet published, much new information and thinking about recovery, particularly at the population scale, was presented. The other was completion of the scientific work by the joint ICES-JRC project to provide the scientific basis for implementation of the Descriptors in the Marine Strategy Framework Directive (MSFD). In the reports of several Task Groups and Management Group Report (Cardoso *et al*., 2010) the need for an interpretation of the concept of "recovery" that is operational and consistent across a wide range of ecosystem components from individual populations to ecosystems is clear. For that reason, WGECO revisited its work on "recovery" looking specifically at the concept in the contexts of populations and ecosystem properties above the level of individual populations, and relative to use of the term in the scientific literature, rather than primarily in policy documents.

Moreover, the relationship between a level of perturbation and the capacity of the ecosystem to recover from it appears to be essential in analysing which level of perturbation can be considered as sustainable, as outlined by WGECO (ICES, 2008) and stressed by WGECO (ICES, 2009). In the present report we examine this relationship within the specific perspective of setting reference points for ecosystem properties.

Section 4.1 reviews descriptions and possible definitions of recovery and related terms at the population and community levels that have been found in the management-oriented literature, agreements and legislation, and past ICES work. It also considers briefly the concept of "resilience" in ecological systems, because many sources have interdependent treatments of resilience and recovery. Section 4.2 summarized the results of 4.1 specifically in the context of other parts of this WGECO Report, where the ability of ecosystems and their components to recover from perturbations is an important part of the science guidance on implementing the Marine Strategy Framework Directive (MSFD). Section 4.3 then discusses the nature of analyses required to examine relationships between perturbations and recovery of ecosystems and their components that would lead to appropriate properties of reference levels of indicators, relative to recovery benchmarks.

4.1 Recovery and resilience

4.1.1 Recovery used for populations

In several reports, publications and other documents analyzed, it was found that the term 'recovery' is used in most cases without further defining what is meant by recovery and to what exact levels the component should recover from or be rebuilt to. The term 'recovery' is sometimes used in a sense that after a decline, any increase in population size could be interpreted as some sort of recovery, for example for overfished fish stocks (Hutchings, 2000). Other sources, however, may stress that even substantial improvements in the status of badly depleted stocks should not be called "recovery', but at most it should be stated that progress *towards* recovery has been made (DFO, 2009).

For some (but not all) fish stocks, distinct thresholds (like B_{lim} or B_{pa}) are used as reference levels or thresholds, for management action. These become *de facto* benchmarks for recovery to the extent that rebuilding plans for stocks below these reference levels often use B_{pa} as the target for rebuilding (e.g., ICES, 2001).

The Annex II of the UN Agreement on Straddling Fish Stocks and Highly Migratory Fish Stocks states: "If a stock falls below a limit reference point or is at risk of falling below such a reference point, conservation and management action should be initiated to facilitate stock recovery." However, no criterion for what constitutes "recovery" is specified in the Fish stocks Agreement.

In its 2009 report, WGECO states that "*In Canada, recovery targets in abundance and range must be specified as part of the mandatory recovery plan for protected species. However despite discussion at two expert meetings, there still is a lack of consensus on how to operationally interpret the term 'recovery' (DFO, 2005; DFO, 2007c). One interpretation is that 'recovery' is reached when a population no longer qualifies under any of the criteria used by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC). These are the same as the IUCN Red List. The other interpretation is that 'recovery' is not reached until a population is at least larger than the limit reference point ('Lower Stock Reference Point' sensu DFO, 2002, 2006c). It is unclear where guidance to resolve this difference of opinion among experts could be found. However, in practical applications it has large implications, both because the two benchmarks (size or range meeting the IUCN/COSEWIC threatened species criteria and limit reference points for populations size) are quite far apart*."

Within the United States, recovery plans are associated with the recovery of critically endangered species from the risk of population extinction whereas rebuilding plans are associated with the recovery of depleted marine capture fisheries and rebuilding the stock to reach more productive levels of exploitation (Wakeford *et al.*, 2007; 2009).

The U.S. Sustainable Fisheries Act requires rebuilding overfished stocks "*to a level consistent with producing maximum sustainable yield*." The Act further requires that the rebuilding time period should not exceed ten years, with exceptions for situations where rebuilding within a decade is not biologically feasible. Guidelines to the Act specify that an "overfished" resource is one that has been depleted to a minimum stock size threshold (e.g., 50% of B_{MSY} for many stocks). A precise translation of this legal text into biological reference points for fisheries is "*MSY is the management strategy, F_{MSY} is the limit reference point, and B_{MSY} is the rebuilding target*." (Brodziak *et al*., 2004).

In the case of marine mammals, ASCOBANS has adopted an interim goal of restoring the population of harbour porpoises in the Baltic Sea to at least 80% of its carrying capacity (ASCOBANS, 2009).

ICES, 2001 stated that "… *F_{pa} and B_{pa} are the thresholds which constrain advice or which likely trigger advice for the implementation of management/recovery plans. If a stock is regarded as depleted, or if overfishing is taking place, the development and effective implementation of a rebuilding plan to reduce fishing mortality to no higher than F_{pa} and to rebuild SSB to above B_{pa}, within a "reasonable" period, would satisfy the condition that management is consistent with a precautionary approach. The word rebuilding appears to be more appropriate than recovery, as it implies that management action is being taken, whereas a recovery could stem from natural causes irrespective of any remedial action"*.

Recently, stock recovery is increasingly recognized as not being synonymous with stock rebuilding. The term recovery tends to be used relatively indiscriminately and often simply denotes recovery of bulk biomass, i.e. stock tonnage. On the other hand, rebuilding should be regarded as a more complex and challenging goal to achieve, aiming to reconstitute a previously evident age-structure which has been truncated by excessive fishing pressure, modified or lost behavioural traits (e.g., the extent and pathways taken during migrations) as a result of altered demography (e.g., communal memory or experiences previously resident in parts of the stock which have been decimated), changed structure of the stock's gene pool and evolutionary mechanisms resulting from diminution of the gene pool arising from substantial depletion or collapse of the stock due to overfishing. Such rebuilding may take generations to achieve, if it can be done at all (Hammer *et al*., in prep.; Murawski, submitted).

4.1.2 Recovery used above the population level

In the FAO Guidelines for Deep-Sea Fisheries (FAO, 2009 a, b) recovery is central to determining whether impacts are "significant adverse impacts" requiring conservation measures to be implemented:

"17. Significant adverse impacts are those that compromise ecosystem integrity (i.e. ecosystem structure or function) in a manner that: (i) impairs the ability of affected populations to replace themselves; (ii) degrades the long-term natural productivity of habitats; or (iii) causes, on more than a temporary basis, significant loss of species richness, habitat or community types. Impacts should be evaluated individually, in combination and cumulatively.

18. When determining the scale and significance of an impact, the following six factors should be considered:

> *i. the intensity or severity of the impact at the specific site being affected;*
> *ii. the spatial extent of the impact relative to the availability of the habitat type affected;*
> *iii. the sensitivity/vulnerability of the ecosystem to the impact;*
> *iv. the ability of an ecosystem to recover from harm, and the rate of such recovery;*
> *v. the extent to which ecosystem functions may be altered by the impact; and*
> *vi. the timing and duration of the impact relative to the period in which a species needs the habitat during one or more of its life-history stages.*

19. Temporary impacts are those that are limited in duration and that allow the particular ecosystem to recover over an acceptable time frame. Such time frames should be decided on a case-by-case basis and should be in the order of 5–20 years, taking into account the specific features of the populations and ecosystems.

20. In determining whether an impact is temporary, both the duration and the frequency at which an impact is repeated should be considered. If the interval between the expected disturbance of a habitat is shorter than the recovery time, the impact should be considered more than temporary. In circumstances of limited information, States and RFMO/As should apply the precautionary approach in their determinations regarding the nature and duration of impacts."

These provisions never define what status the parts of the ecosystem that have been impacted must be in to be considered "recovered". However, they provide substantial guidance on the types of considerations that apply when evaluating recovery, including maintenance of ecosystem structure and function, the ability of populations to increase, the productivity of habitats, and species richness and habitat diversity.

According to definitions given by the Society for Ecological Restoration, 2004, ecological restoration is the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed. An ecosystem has recovered - and is restored - when it contains sufficient biotic and abiotic resources to continue its development without further assistance or subsidy. It will sustain itself structurally and functionally. It will demonstrate resilience to normal ranges of environmental stress and disturbance. It will interact with contiguous ecosystems in terms of biotic and abiotic flows and cultural interactions.

Elliot *et al*. (2007) revised the concept of restoration in estuarine, coastal and marine ecosystem and provided an overview of the recovery terminology. According to their paper, recovery is divided into recovery with (active) and without (passive) human intervention.

Passive recovery pertains to the recovery that *"will occur in ecosystems once stressors have been removed. This depends on properties allowing them to either absorb change or attain an improved structure and functioning. These properties include recoverability, resilience and adaptation but also carrying capacity as an indication of the overall desired state of the system"*.

In this context, they define recoverability as '*the ability of a habitat, community or individual (or individual colony) of species to redress damage sustained as a result of an external factor'* (MarLIN Glossary, 2005).

At the same time different definitions are provided either for resilience and resistance, which are summarized as follows: *"resistance and resilience are inherent properties of the ecosystem which indicates its ability to absorb change against a background of the complexity and/or variability of the ecosystem"*. The interpretation of these concepts is well illustrated in Figure 4.1.2.1. The authors provide a conceptual model of changes in the state of a system with increasing pressure. Given any functional parameter, resistance is defined as the amount of pressure that can be applied without deterioration in its status. As pressure is removed, Type I Hysteresis represents the lag in recovery, i.e., status may not improve for some time after the pressure has been removed. Accordingly Type II Hysteresis represents the difference between the original status and the status achieved after the release of the pressure, being a measure of how resilience is impaired due to the effects of disturbance.

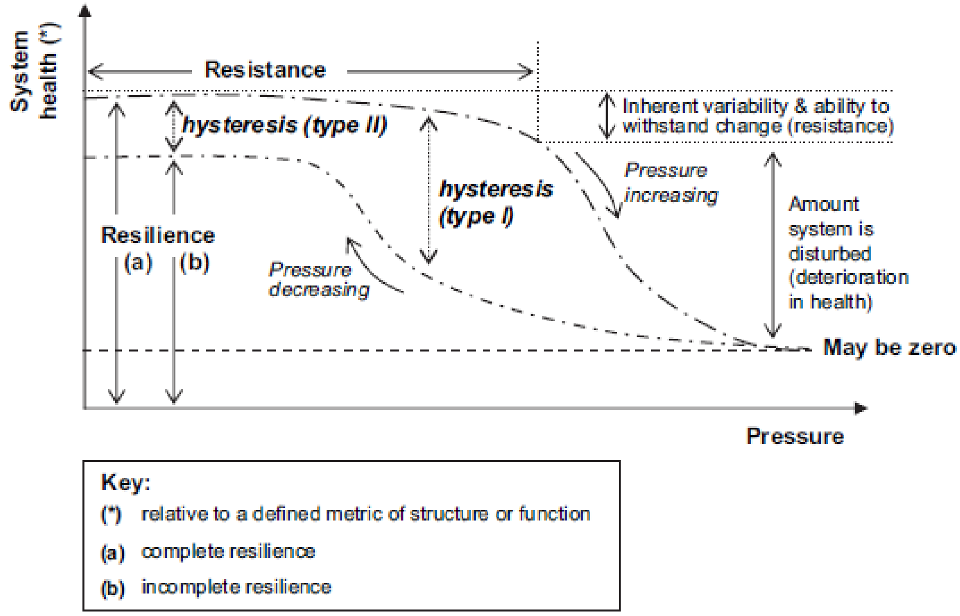


Figure 4.1.2.1. A conceptual model of changes to the state of a system with increasing pressure (Modified from Elliot *et al.*, 2007). Hysteresis refers to systems which have memory; that is, the effects of the current input to the system are not felt at the same instant.

When dealing with **active recovery**, the authors refer to *"human-mediated actions used to enhance recovery (...). These have been classified into actions combating a degraded environment and the effects of a single stressor"*. Thus, active recovery lies in the domain of restoration activities, including for instance rehabilitation, remediation and re-creation, re-introduction, re-establishment, reclamation and replacement (see Elliot *et al.*, 2007 for details).

According to the above definitions, Elliot *et al.* (2007) provide a conceptual model illustrating the nature of natural recovery of a degraded habitat (Figure 4.1.2.2).

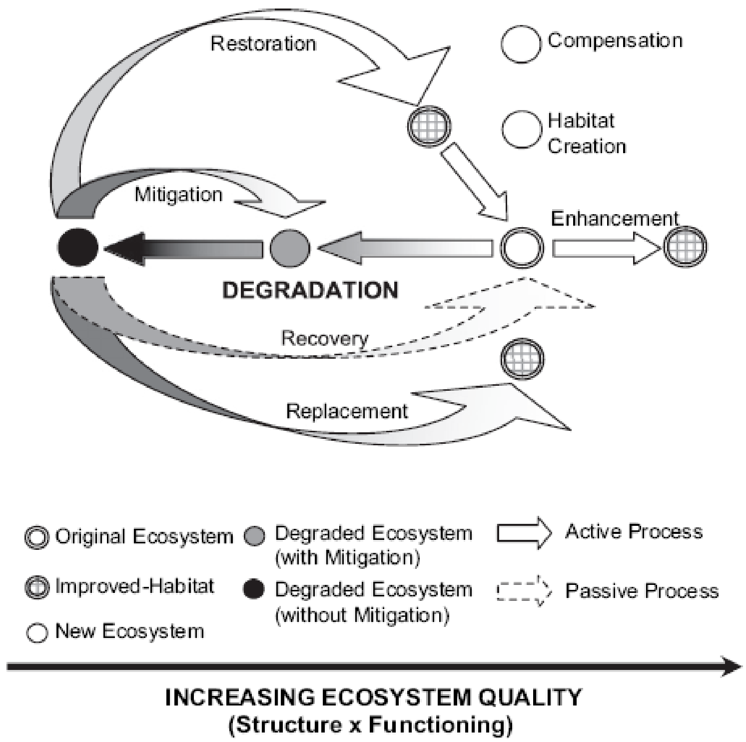


Figure 4.1.2.2. A conceptual model illustrating the nature of natural recovery of a degraded ecosystem and the terms used in human-mediated (active) restoration. The model indicates that habitats can be produced which are an improvement on the degraded state but not necessarily to the original state, whereas other ecosystems are newly created systems. The recovery (light grey dashed arrow) can be to the original state or some distance along that pathway of regaining ecosystem quality. The model emphasises the movement of ecosystems along a continuum (horizontal axis) of ecosystem quality, which combines both structure and functioning, whereas the position of ecosystems in the vertical axis in the model has no meaning (Modified from Elliot *et al.*, 2007).

According to Borja and Elliot (2007) there are many definitions of what is **good ecological restoration**, and they give as one of the most widely cited definitions ''*...restoration is defined as the return of an ecosystem to a close approximation of its condition prior to disturbance. In restoration, ecological damage to the resource is repaired. Both the structure and the functions of the ecosystem are recreated. Merely recreating the form without the functions, or the functions in an artificial configuration bearing little resemblance to a natural resource, does not constitute restoration. The goal is to emulate a natural, functioning, self-regulating system that is integrated with the ecological landscape in which it occurs. Often, natural resource restoration requires one of the following approaches: reconstruction of the antecedent physical, hydrologic and morphologic conditions; chemical cleanup or adjustment of the environment; and biological manipulation, including revegetation and the reintroduction of absent or currently nonviable native species*''.

However, there is also the view that "*Ecosystems are naturally variable, so even a successful recovery program will not return an ecosystem to exactly the state is was in prior to the perturbation. What point constitutes recovery – presence or maturity?*" (FAO, 2009 a, b). And Mee *et al.* (2008) ask "*However, this raises two questions: how to determine a baseline or degree of recovery in a highly mobile system, such as mobile sands and gravel on the seabed, which are developed by having constant reworking of their sediments, and secondly what should systems recover to? Two centuries ago the English Channel had extensive oyster beds –*

a completely different habitat than any current one. Oyster beds were destroyed by overexploitation and pollution in the 19th Century but, at that time, the more mobile flatfish flourished. Since then, the entire area has been subjected to heavy trawling, another major source of impact, and flatfish populations have dwindled. Should a baseline be a seafloor abundant in oysters or one having large populations of flatfish?"

The need of defining historical baselines to be compared to the present status of marine resources and ecosystems was highlighted by Pauly (1995), who argued that the speed of changes in biodiversity is so high that researchers belonging to different generations have quite different perceptions of the "natural" baseline of ecosystems and their components.

This analysis fostered the development of multidisciplinary research effort in the framework of marine historical ecology, aimed to determining long-term changes and baselines (i.e., up to centuries, thousands of years BP) in marine animal populations, communities and ecosystems, by integrating historical and archaeological sources, naturalists' reports, traditional ecological knowledge, zoo-archaeological remains, paleoecological data with grey literature documents, scientific and statistical data (Jackson *et al*., 2001; Rosenberg *et al*., 2005; Saenz-Arroyo *et al*., 2005; Lotze *et al*., 2006; Fortibuoni *et al*., 2008; Pinnegar *et al*., 2008; Lotze and Worm, 2009).

The intrinsic variability of ecosystems, being dynamic and complex systems, is determined by the combination of natural variability and the effects of human pressures, because humans played an important role in shaping ecosystem structure and functioning even in the past. To capture this concept, in the framework of the historical ecology discipline, several definitions have been proposed so far, including: *range of natural variation* (Caraher *et al*., 1992), *natural variability* (Swanson *et al*., 1994), *reference variation* (Manley *et al*., 1995), *ecosystem of reference* (Aronson *et al*., 1993) and *historic range of variation* (Morgan *et al*., 1994; Aplet and Keeton, 1999). Only the latter definition takes into adequate consideration the role of human pressures. Moreover it avoids the use of the term "natural" that would be, as a consequence, misleading. From the conceptual point of view, the historic range of variation, by taking into account the dynamic and complex behaviour of ecosystems, points to the definition of the range within which they were self-sustaining (given the historical human pressure) and beyond which they move into an unsustainable state.

Turnhout *et al*. (2007) suggest that ecological indicators, although they are highly dependent on scientific knowledge, cannot be solely science-based, due to the complexity of ecosystems and the normative aspects involved in assessing ecosystem quality. According to Foden *et al.* (2009), physical recovery (TPhys) and biological recovery (TBio) were determined as the mean time-period for recovery to pre-dredge or reference site conditions.

4.1.3 Resilience

WGECO (ICES, 2009) defined two types of resilience in the twelve sources examined. These were based on either the **'resistance' of an ecosystem to stress** (e.g., the ability of an ecosystem to '*maintain its structure and pattern of behaviour in the presence of stress*' (FAO, http://www.fao.org/docrep/005/y4470e/y4470e0h.htm), or on **an ecosystem's 'response' to disturbance** (e.g., the IUCN 2003 definition used by OSPAR 2006 which was, '*the ability of an ecosystem to recover from disturbances within a reasonable time frame*').

Hughes *et al*. (2005) defined resilience as "*the extent to which ecosystems can absorb recurrent natural and human perturbations and continue to regenerate without slowly degrading or unexpectedly flipping into alternate states*."

Finally Levin *et al*. (2008) summarized work by the Resilience Alliance (http://resalliance.org), which makes a distinction between **engineering resilience** (namely, "*the rate at which a system returns to a single steady or cyclic state following a perturbation*") and **ecological resilience** (namely, "*the amount of change or disruption that is required to transform a system from being maintained by one set of mutually reinforcing processes and structures to a different set of processes and structures*"): "*It is clear that the notion of resilience is sometimes interpreted in the general literature in the narrower sense of recovery from disturbance, and at other times in the broader sense of the maintenance of functioning in the face of disturbance. For the remainder of this article we use the terms robustness and resilience interchangeably to mean* ***the capacity of a system to absorb stresses and continue functioning***." (Levin *et al*., 2008).

Levin *et al*. (2008) then go on to note that "*central themes in management, and throughout this special section, are the conditions under which robustness and resilience may be lost as a result of endogenous or exogenous influences***.** *The dominant paradigm discussed here, borrowed from catastrophe theory (Thom, 1975), is of a dynamical system characterized by multiple basins of attraction at any given point in time. Over fast timescales, such systems may be expected to approach (possibly dynamic) asymptotic states of lower complexity and dimensionality than the transient dynamics; over longer (slower) timescales, the shape of the dynamic landscape changes, and the stability of those asymptotic states may be compromised. The result may eventually, over even longer timescales, be a transition to a new asymptotic state. The changes may be subtle: erosion of adaptive capacity or buffering from the loss of biological diversity, say, may expose the system to the effects of novel perturbations, but the consequences may take a while to appear*."

4.1.4 Conclusions from review of definitions

A partial tabulation of information (Table 4.1.4.1a and b below) from the single population scale illustrates that the use of additional terms can bring some order to the application of these concepts in science, policy and management. However, even when multiple terms are used to partition complex aspects of the concepts, different sources break down the related concepts in varying ways.

Table 4.1.4.1. a) Definitions of the terms 'recovery' and 'rebuilding' of (single) populations.

	Term	
Source	Recovery	Rebuilding
ICES, 2001	Process (natural) of the overfished stock/population; could stem from natural causes	Management action being taken to reduce F below Fpa and SSB above Bpa
UNCOVER Project (Hammer *et al*., in prep), Murawski, submitted	Increase of stock level (above a certain threshold)	Achieving previous life history traits, like age structure or migration routes
U.S. legislation (as in Wakeford *et al*., 2007)	Associated with critically endangered species	Associated with depleted marine capture fisheries
Hutchings, 2000	Any increase in population size	
DFO, 2009	Even substantial improvements in the status of badly depleted stocks should not be called, "recovery', but at most it should be stated that progress towards recovery has been made	

Table 4.1.4.1. b) Definitions used for recovery/restoration for ecosystems.

Source	Definition
SER, 2004	An ecosystem has recovered - and is restored – when: It contains sufficient biotic and abiotic resources to continue its development without further assistance or subsidy. It will sustain itself structurally and functionally. It will demonstrate resilience to normal ranges of environmental stress and disturbance. It will interact with contiguous ecosystems in terms of biotic and abiotic flows and cultural interactions.
Borja and Elliot, 2007	Restoration is defined as the return of an ecosystem to a close approximation of its condition prior to disturbance.
Foden *et al*., 2009	Habitat to pre-dredged or reference site conditions.

When the concept of recovery has been explored above the species level, things get no more consistent. However, from the various sources, some common messages emerge. All sources stress that the central consideration is **maintenance of ecosystem structure and function in conditions characteristic of the ecosystem** in some past time. This message is consistent, whether discussed at the levels of populations, communities, or habitats. Section 3 of this report contains much relevant information on how to evaluate the status of ecosystem structure and function, and how to select the key aspects of ecosystem structure and function for a given system. Specifically with regard to "recovery" of ecosystem structure and function, several sources also highlight that recovery does not require being all the way to whatever previous state has been chosen as a target.

4.2 Recovery in the context of this Report and the MSFD

Based on the literature review, WGECO consider that these five characteristics are the most important:

1) the necessary pieces for "normal" structure and function are present, and

2) further progress towards that target is likely without any special management measures incremental to those expected for sustainable use.

Likewise the population, community, habitat or ecosystem has at least regained sufficient resilience such that:

3) perturbations greater than those associated with the normal range of environmental forcing and/or sustainable use would be necessary to cause impairment of the structures, functions, and processes associated with the population, community, habitat or ecosystem.

There is also a consistent message about a few things that are not part of operational definitions of "recovery".

4) It is not necessary for every single species historically observed in a community or ecosystem to be present, as long as those species or functional groups necessary for normal structure and function are present.

5) Likewise the historical abundances, biomasses, age compositions, etc. of all the species or functional groups noted in 4) do not have to be at historical levels, as long as further progress in the direction of recovery is considered secure under "normal" management.

Those conditions are an operational interpretation of how "recovery" could be evaluated with ecological consistency for populations, habitats communities, and ecosystems. States may choose to set higher standards as their goals for management and recovery. However, at least these standards must be met for "recovery" to be consistent with the intent of the diverse policy instruments reviewed in 2009, and the scientific literature reviewed this year.

The terms 'recovery' or 'rebuilding' are not used in the MSFD, the MSFD aims to "*… where practicable, restore marine ecosystems in areas where they have been adversely affected*". In Annex V of the MSFD, it is stated that monitoring programmes "*need to include … the possible corrective measures that would need to be taken to restore the good environmental status, when deviations from the desired status range have been identified*." This is the only place where the MSFD gives a concrete target (GES) for the restoration activities called for. In the preamble, it is stated that the marine environment "*must be … where practicable, restored with the ultimate aim of maintaining biodiversity and providing diverse and dynamic oceans and seas which are clean, healthy and productive*."

On the other hand, a number of the ICES/JRC Task Groups working on the science basis for implementation of the individual descriptors considered "recovery" to be a central consideration in setting standards for GES. This was particularly the case for the Descriptors of Biodiversity, Alien Species, Food Webs, and Seafloor Integrity (see Table 6.3.3.1.1 of this Report). In each case the concept of "recovery" was used in the context of determining if an impact was or was not sustainable. Given its important role in the proposals for implementation of the MSFD, Section 4.3 considers how to make the concept of "recovery" actually operational in the context of sustainability.

4.3 Analyses required to examine the relationships between perturbation and recovery

4.3.1 Conceptual framework

When setting reference levels that should reflect the policy objective "sustainable use", it is necessary to apply a line of consistent ecological reasoning about what level of alteration of the attributes being measured by the indicator is *not* sustainable, and set the reference level to avoid that level of alteration. As summarized in past WGECO reports, there has been substantial scientific debate about appropriate benchmarks for the boundary between sustainable and unsustainable use, and the appropriate ways to deal with uncertainty and natural variation in this boundary condition (ICES, 2006; 2008). The reasoning was developed most fully in building the fisheries advisory frameworks, and even though ICES is changing that framework to accommodate a new EU policy objective for fisheries, the reasoning is a useful guide to setting reference levels associated with sustainable use. Additional guidance on what standards are being accepted by both science and policy communities as benchmarks for sustainability can be found in the work of higher-level intergovernmental marine agencies such as FAO and CBD (FAO, 2008; CBD, 2008).

Evaluations of the degree to which perturbations are sustainable always have to consider at least two factors; the degree to which recovery of ecosystem attribute from the perturbation is rapid and secure, and the degree to which functions served by the ecosystem attribute and ecosystem processes in which the attribute plays a key role are altered. Impacts cease to be sustainable when ability to recover or serve important ecosystem functions is impaired. Section 4.1 discusses the concept of recovery in some detail, both for populations and ecosystem attributes above the scale of populations. Section 3.5 provides advice on important ecosystem functions and processes, and how to associate a wide range of types of indicators with those functions and processes.

The rest of this Section uses that information in developing lines of ecological reasoning that can help to position reference levels for sustainable impacts on indicators. There are often several ecological standards that can be applied in seeking to set a reference level. Sometimes there is a logical order in which these standards might be attempted, and the reference level would be the value of the first standard that is met. In other cases all the standards should be considered, and the reference level is the highest level (that is, the level associated with the least altered status on any of the standards).

4.3.1.1 Capacity to recover

For indicators of ecosystem attributes that have the capacity to recover from perturbation, it is always necessary to evaluate the point at which such recovery is no longer likely to be rapid or secure. "Rapid" is always interpreted relative to the life history parameters of the population of concern; rapid for a small pelagic is not the same rapid for a large cetacean. "Secure" is interpreted relative to the likelihood that recovery would start immediately were the pressures causing the mortality reduced, and that in its current status, the population is not at increased risk of major further losses ue to stochastic factors and likely scales of natural pressures.

For individual populations, recovery potential is almost always evaluated by taking some measure of the population's ability to produce recruits. In fish stock assessments this has traditionally been done by looking at how recruitment has varied with mature biomass. If there appears to be some functional relationship between recruitment and stock (Beverton–Holt, Ricker, "hockey-stick", etc.) that is used to estimate the mature biomass below which the expected recruitment is not large enough to replace the current biomass and allow biomass to increase "quickly" (given the life history of the species) if fishing pressure were reduced. This is usually in the region of the ascending limb of the functional stock-recruit curve where the slope changes the most (ICES, 2007); above this point the proportional rate of change in expected recruits is slower than the proportional rate of change in mature biomass. Below this point expected recruits are lost at a faster relative rate than mature biomass.

When data are insufficient to fit a functional stock-recruit curve, or show no evidence of a functional relationship over the range of data available, a variety of alternatives have been used successfully to estimate the biomass associated with impaired recruitment. These include:

- Probabilistic methods for estimating how the likelihood of a recruitment as low or lower than one which, if experienced on average would not allow a population to have rapid secure recovery, varies with mature biomass.
- Reviewing historical population data and identifying the lowest historical size from which rapid and secure recovery was observed.

In no sense is this general approach restricted to just estimates of mature biomass and estimates of numbers of recruits. Any measure of population status and any measure of the ability of the population to replace itself can be used as independent and dependent variables (the "x- and y- axes", respectively). The biological interpretation of the functional relationship (or other method of evaluating dependence) will have to respect the nature of the indicators used to measure stock status and stock productivity, but the concept of seeking some point in the relationship can always serve as a consistent standard for identifying the position on the population indicator below which its ability to replace itself is at risk of being impaired.

For ecosystem attributes that are not populations but still have the capacity to recover from perturbations, WGECO has already argued that the same general approach can be followed to estimate reference levels associated with impairment of capacity to recover (ICES, 2008), and Rice, 2009 developed the methods to operationalize that argument more fully. An example of such an attribute might be "biomass at trophic level 3". This is not a population of a species, but an aggregate property of many species in a food web. Nonetheless, it is ecologically straightforward to build an argument that the status of biomass at trophic level 3 sometime in future may depend on the biomass of trophic level 3 now. It is also the case the biomass at trophic level 3 in the future depends on many other things as well (just as future SSB depends on many factors other than current recruitment). However, impairment of biomass at trophic level 3 now can certainly mean that any recovery of biomass at trophic level 3 in the future may not be secure, or take much longer to achieve (just as impairment of current recruitment through depletion of SSB makes it less likely to reach recovery goals rapidly and securely).

For all ecosystem attributes with a capacity to recover from perturbations, to estimate the level of the indicator below which replacement likelihood or ability is impaired one needs an ecological rationale for why the "x-axis" indicator reflects the status of the ecological attribute of concern, and why the "y-axis" indicator reflects the potential of that ecological attribute to increase. For a parallel, there is usually a good ecological rational for using SSB to reflect the status of an exploited fish stock and R to reflect the potential of the stock to increase in future. And just as with SSB and R, there are many complications that make this simplification imperfect, but despite the imperfections, it is a simplification that often is enough for fisheries management decisions to be made. Nonetheless, the ecological argument for "y-axis" attribute as some factor in the ability of the "x-axis" attribute to recover is a key part of setting reference levels for any ecosystem attribute with the capacity to recover from perturbations.

Where there does seem to be some non-linearity in the functional relationship between the "x-axis indicator" and the "y-axis indicator" the methods in WGECO (ICES, 2008) and Rice, 2009 can be used to estimate a reference level for sustainable impacts on the attribute represented by the x-axis indicator. In cases when there is no change in the "y-axis" indicator over the full range of the "x-axis" indicator for which information exists, then the data are saying that there is no level of the ecosystem attribute for which the "x-axis" is an indicator below which recovery ability is impaired. This suggests in turn that all levels of reduction of the ecological attribute are sustainable, with regard to ability to recover. Then it is necessary to move to the evaluation of ecosystem function. If there is a strictly linear relationship between the x-axis indicator and the y-axis indicator over the full range of the "x-axis" indicator for which information exists, then the data are saying that the ability of the population (or other ecosystem property) to recover from perturbations is directly proportional to population size (or x-axis value more generally), and the possibility of recovery remains until the population becomes so small that stochastic factors pose a threat to the existing population. Again it is necessary to move to the evaluation of ecosystem function.

4.3.1.2 Capacity to serve ecosystem functions

For ecosystem attributes which have no capacity to recover from perturbations, reference levels for sustainable impact have to be set based on impairment of the functions served by those attributes. For example the three-dimensional structure of seafloor habitat or the amount of gravel in the seabed may have no capacity to recover from damage or removal. That does not mean that from an ecosystem perspective there is no use or impact on those features that is sustainable. Rather, evaluations of the sustainability of the reduction on these attributes will be based on the functions provided by those attributes.

This question of what functions are served by various ecosystem features and how the features contribute to ecosystem processes covers the entire field of ecology. Table 3.5.4.1.1 gives some entry to this vast field, providing a synopsis of ecological function and process served sometimes or usually by the classes of indicators in the MSFD Elements of a Decision. For each individual case when a reference level for sustainable impact on an indicator of an ecosystem function must be set, two ecological cases must be presented and justified. The first is that the ecological attribute of concern really does serve the function of concern. The second is that the indicators used for the "x-axis" and "y-axis" respectively really reflects the status of the ecological attribute of interest and degree to which the function is being served. (Both of these cases should be made and justified whenever the indicators are chosen for any use, so this is not a new and unique demand for setting reference levels. In addition, as practice develops, documenting these justifications provides information for wider application. This increases efficiency and consistency of practice, and engages the science community in challenging and validating the justifications.)

Once the "x-axis" and "y-axis" indicators of the status of the ecosystem attribute of concern and the function it serves are selected and justified, the process proceeds just like for setting reference levels for capacity to recover. The only differences are that the interpretation is about how the function served by the attribute varies with an appropriate measure of the quantity or quality of the attribute. "Impairment" or "harm that is serious and difficult to reverse" to a function would be considered to occur when the level of the function fell to a point where parts of the ecosystem depending on that process lost their own ability to recover, or declined themselves to levels where their own functions in the ecosystem might be impaired.

For example, if the ecological attribute of interest was amount of macroalgae (that could be impacted by many human-induced pressures) and the function was provision of nursery habitat for fish, then the relationship to be interpreted might be how an index of recruitment for the fish thought most dependent on macroalgae for shelter varied with the number of hectares of seagrass or kelp. An unsustainable impact on macroalgae would when it was reduced to a level where recruitment of the dependent fish was impaired (evaluated using the population-based approaches described in the Section above).

Given our incomplete knowledge of ecosystem structure, function, and processes, it will often be the case that these relationships are data-driven (when data are available) and correlative, and the full process-based understanding of the linkages between the ecosystem attribute and the functions it may serve is not available. Again, however, this will be true regardless of how indicators of ecosystem function are to be used in assessment and associated with reference levels. When data are available to explore the relationship between the indicator of status of the ecosystem attribute and the indicator of its function, the first step is to seek evidence of non-linearities in the relationship. Again, this might be done through estimating parameters of a hypothesized functional relationship, probabilistically estimating the likelihood of a level of the function so low that some part of the ecosystem dependent on the function suffered impairment or serious harm, or other methods that may be developed in future. When there is no evidence of non-linearities, the next step is to consider other functions that may be served by the same ecosystem attributes. In the end, the only situation when there would be no ecological basis for setting reference levels for ecosystem attributes based on their functions would be when one of two conditions is met:

- It is not possible to find any function served by the ecosystem attribute that declines as the quantity or quality of the ecosystem attribute declines.
- It is not possible to find any function provided by an ecosystem attribute, that, if it were reduced, would result in some population or ecosystem component suffering impairment of it own productivity, or of its ability to serve its own functions in the ecosystem.

This framework could be made into an infinitely recursive process of layering function on function, and attribute on attribute. However, the type of ecological rationales used creating Section 3.7, if applied systematically, should guide science efforts to address higher priority ecological issues ahead of lower priority issues.

4.3.1.3 Complications

The status of the ecosystem attribute just inside the reference level may still be highly altered from its unimpacted state, and society may choose to have a less impacted status as a management target. However, that does not change the value of the reference level on the indicator that is associated with *sustainable use.* It just adds another reference level for a status where impacts are less than those associated with maximum sustainable use.

There may be time lags between the alteration of an ecosystem attribute and the evidence that an ecosystem function served by the attribute has been degraded. These lags need to be taken into account, but are often hard to quantify.

The spatial scales of the indicator of changes in an ecosystem attribute and the indicator of the function served by the attribute may be hard to align. For example, impacts on nursery habitats may be measured (and occur) on much more local scales than the most readily available indicators of recruitment to the species using the habitat, so a more appropriate indicator of the function may have to be sought.

4.3.1.4 How to deal with pressures in this framework

This framework is designed to estimate reference levels for state indicators; whether the state of populations, communities or habitats. Because the first-order policy and management questions are likely to be about what level of a pressure is sustainable, it is essential that pressures be integrated into this framework for setting reference levels.

It will often be easy to develop plots of how the status of an ecosystem attribute will vary with the level of a pressure that is applied, whether from data or from theory. These plots too may be non-linear, but they have some important differences from the plots discussed above of how an ecosystem function or ability to recover varies with the quantity or quality of an ecosystem attribute. First, the curvature will be more often concave rather than convex; the ecosystem attribute is expected to be high when the pressure is at zero, and drop rapidly as the pressure begins to impact the ecosystem attribute (whereas recruitment is expected to be very low at low SSB, and increase when SSB begins to increase). More importantly there is no information in that single plot which gives information about the ecological consequences of the reductions in the ecological attribute. For example, in a classical plot of how stock biomass varies with F (or effort), the relationship is strongly concave [For example, the curvature in Figure 4.1.2.1]. However, for any desired F or effort, there is some equilibrium stock biomass – just one that is smaller and smaller as the consistently applied F increases. There is nothing *in that plot* that makes any one outcome any better or worse than any other, and hence provides no basis for a reference level for F. To find a reference level for F, it is necessary to consider the ecological implications of different levels on the stock biomass axis.

Fortunately, the framework just summarized above provides a relevant and robust way to consider those ecological implications. One can ask directly, using the population recovery steps, how does ability of the stock biomass to replace itself decline with stock size? If a reference value for impaired productivity of the stock can be found using the population recovery approach, then it determines that stock biomass should not fall below that level. One then uses the stock – F relationship to identify the F that implies an equilibrium biomass that meets the criterion for a reference level for stock biomass. That F becomes the reference level for F, but used in reverse. Just as sustainable use must keep biomass *above* the biomass reference level; it must keep the F *below* the F reference level.

The same approach can bring all pressures into this overall framework for setting reference levels. One needs some knowledge of which ecosystem attributes will be impacted by various pressures, but this information is available in, for example the tables in the Task Group reports and the Management Group report. One also needs some knowledge of how the impact on the ecosystem attribute varies with the intensity of the pressure. Then one must go through the process for population and community indicators of determining if there is some ecological basis for setting a reference level of the population or community indicator, based on its ability to recover and/or the functions it serves in the ecosystem. Once a reference level is identified (and justified) the mapping onto the level of the pressure associated with that level of ecosystem attribute is direct (taking due account, of course, of uncertainties and need for precaution).

4.3.2 Case-history examples

Here we provide case-history examples of the relationships between perturbation and recovery at the population, habitat, and community level. In each case, the x-axis is the measure of perturbation, which is generally related to a pressure indicator. The *y*-axis is a measure of the recovery capacity, which ideally is a rate with units of inverse time, but may also be a state indicator that is a proxy for recovery capacity. The relationship between recovery capacity and the perturbation can be fit with a generalized additive model to determine the shape of the functional form. If the relationship is curved and/or has inflection points, it may be possible to identify standards for recovery.

4.3.2.1 Populations

Seabirds have frequently been used as indicators of the state of the marine food webs that support them (Furness and Greenwood, 1993; Monaghan *et al*., 1989; Harris and Wanless, 1990; Furness and Camphuysen, 1997). In the northwestern North Sea, breeding kittiwakes *Rissa tridactyla* feed primarily on, and provision their chicks with, sandeels *Ammodytes marinus* (Harris and Wanless, 1997; Furness and Tasker, 2000; Lewis *et al*., 2001). Consequently, "Seabirds" is one of the Issues of Ecological Quality identified by OSPAR for which Ecological Quality Objectives (EcoQO) were intended to be set. A study undertaken off the Firth of Forth in southeast Scotland provides the opportunity to examine this EcoQO. Approximately 40 km offshore of an important kittiwake colony, the Isle of May in the Firth of Forth, lies a complex of sandbanks, most notably the Wee Bankie, where a major sandeel fishery started in 1990. The fishery peaked during the early 1990s before being closed in 2000 following concern over the impact of the fishery on sandeel supplies to marine top predators. In this example, the perturbation is the level of sandeel landings and chick production is a direct measure of the capacity of the kittiwake colony to recover. Kittiwake breeding success was significantly (R^2=0.293, P<0.01) negatively related to the sandeel landings (Figure 4.3.2.1.1).

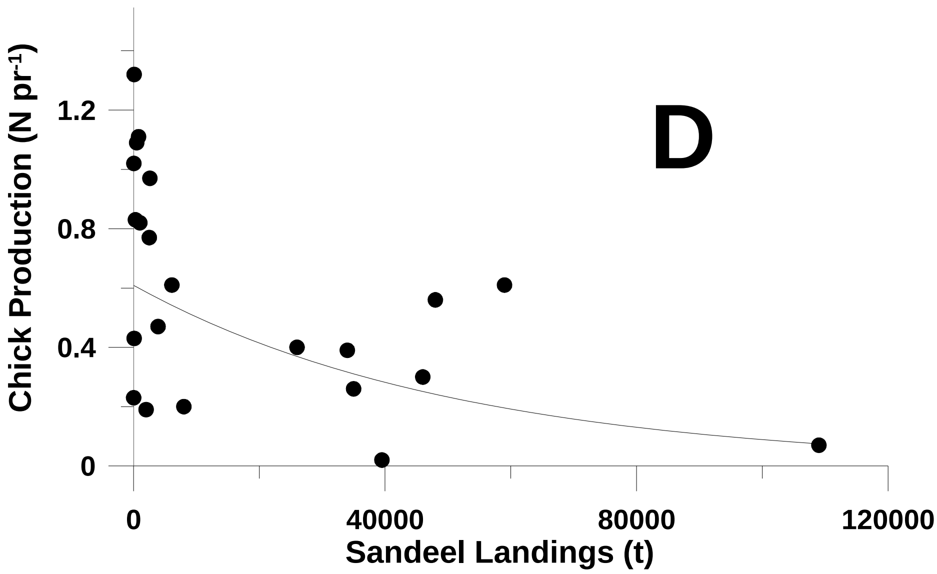


Figure 4.3.2.1.1. Relationship between kittiwake breeding success and sandeel landings.

4.3.2.2 Habitats to be completed by more explanations on habitat

A management experiment was conducted on the northwest Australian shelf, in which two large areas were closed to pair trawling (Sainsbury *et al*., 1997). The proportion of the seabed with benthos increased in the closed areas and continued to decrease with time in the open areas. Large sponges, in particular, provide habitat structure for many invertebrate and fish species. The results of this experiment can be interpreted in the context of perturbation and recovery dynamics. In this example the proportion of the seabed with large benthos is an inverse measure of the habitat damage caused by trawling the seafloor and the recovery capacity is the catch rate of the fish *Lethrinus* (Emperor fish) and *Lutjanus* (Snapper) (Figure 4.3.2.2.1). The catch rate is a more-or-less linear function of the proportion of the sea floor covered with large benthos.

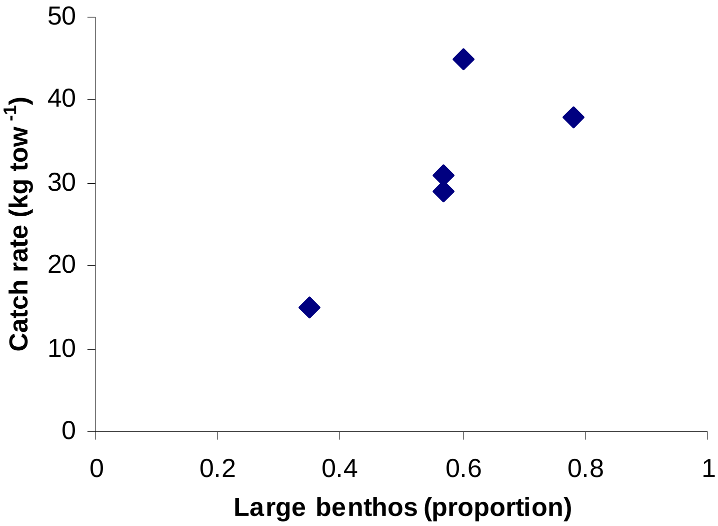


Figure 4.3.2.2.1. Total catch rate of *Lethrinus* (Emperor fish) and *Lutjanus* (Snapper) versus proportions of seabed with large benthos in the zone closed to trawling.

4.3.2.3 Benthic communities

This example concerns the effects of bottom trawling on the benthic fauna of the North Sea (Jennings *et al.*, 2001). The measure of perturbation is an index of trawling disturbance derived from fishing location data. The recovery capacity is the rate of infaunal production, in the sense that a community with higher production will recover more quickly. The relationship between production and trawling disturbance appears more-or-less linear and declines to almost zero at high levels of disturbance (Figure 4.3.2.3.1). The P:B ratio increases with trawling disturbance, because the species that remain in a disturbed community are smaller individuals and species with higher turnover rates. The increase in P:B does not compensate for the decrease in biomass.

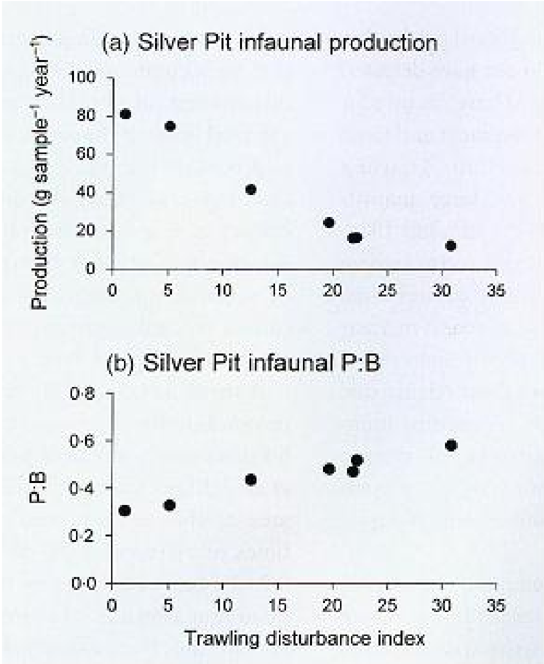


Figure 4.3.2.3.1. Production (a) and production-to-biomass (b) as a function of trawling disturbance at the Silver Pitt site in the North Sea (Jennings *et al.*, 2001).

4.3.2.4 Fish communities

Model results can be used to examine the relationships between fishing perturbation and the recovery capacity of fish assemblages. For example, size-based models have been used to investigate the impact of fishing mortality on several community indicators. The perturbation is the fishing mortality, applied across the community. One measure of recovery capacity is the size diversity of the community, in the sense that populations with a greater diversity of sizes are expected to have a higher proportion of mature individuals. Also, a larger diversity of sizes within the mature component of the population is expected to increase reproductive success and subsequent survival of the offspring. In this example, size diversity decreased with fishing mortality, but the shape of the relationship depended on the size selectivity of the fishing gear (Figure 4.3.2.4.1). When all sizes were fished, the curve was convex upwards; with moderate size selection the relationship was almost linear; with strong selection for large sizes, the curve became concave upwards. This example illustrates that the relationship between recovery capacity and the perturbation depends not only on the magnitude of the perturbation, but also how the perturbation affects different components of the community.

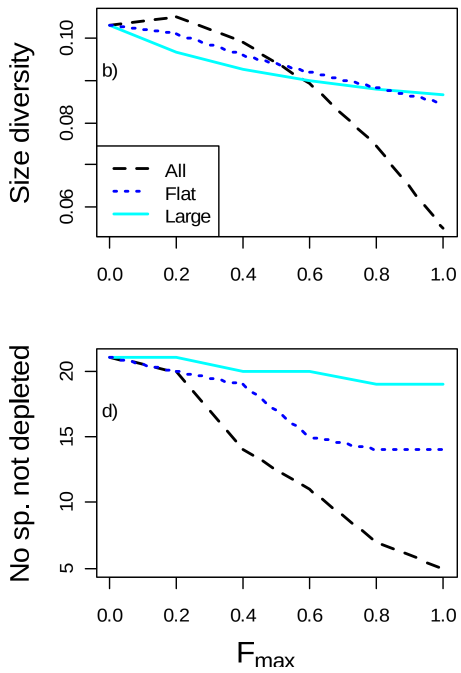


Figure 4.3.2.4.1. Size diversity (b) and the number of species not depleted (d) in relation to the fishing mortality rate on fully-recruited size classes. The size-based model was parameterized for 21 species in the Georges Bank fish community. In this scenario, the fishing gear is assumed to have a logistic selectivity that selects all sizes (All), is a gently increasing function of size (Flat), or selects only large fish (Large). From Rochet *et al.*, in preparation.

The same size-based model (LeMANS) was used to calculate the relation between a number of community indicators and exploitation rate (Figure 4.3.2.4.2). Among these indicators, mean L_{max} is often used as a proxy for size composition in the community. Though not a direct measure of recovery capacity, the reduction in mean size with increased exploitation is likely to affect ecosystem function and to decrease the recovery capacity for the same reason as a decrease in the size diversity will (see above). In this example there are inflection points in the mean L_{max} curve, which could be used to identify threshold levels of exploitation.

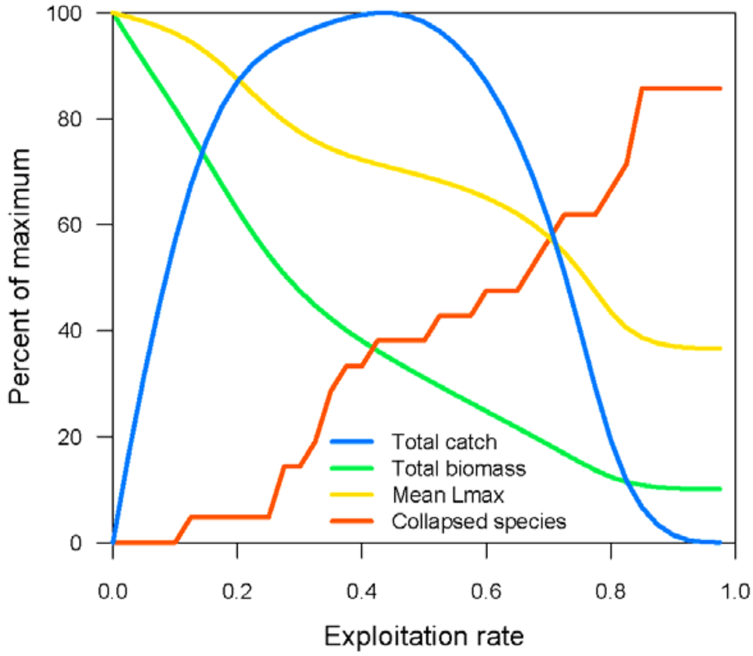


Figure 4.3.2.4.2. Effects of increasing exploitation rate on a model fish community. Exploitation rate is the proportion of available fish biomass caught in each year. Mean Lmax refers to the average maximum length that species in the community can attain. Collapsed species are those for which stock biomass has declined to less than 10% of their unfished biomass. This size-structured model was parameterized for 19 target and two non-target species in the Georges Bank fish community. It includes size-dependent growth, maturation, predation, and fishing. Rebuilding can occur to the left, overfishing to the right of the point of maximum catch. From Worm *et al.*, 2009.

4.3.2.5 Summary of case-history examples

Above the population level (habitats, communities, assemblages) it is possible to identify appropriate indicators of perturbation (x-axis) and recovery capacity (y-axis) although the y-axis is often a proxy for recovery capacity. Section 3.7 provides a complete list of indicators, some of which could serve as suitable proxies. The functional relationships between recovery capacity and perturbation are generally uncertain and may only be known qualitatively (e.g., increasing, decreasing, no change). Although it is informative to consider the relationships between pairs of indicators, it must be remembered that other factors are involved. In the kittiwake-sandeel example, other factors are known to affect kittiwake breeding success, just as factors in addition to fishing affect sandeel availability. Even when the functional relationship can be quantified, there may be no obvious level of perturbation at which the ability to recover is impaired. In these cases, it becomes necessary to consider the links between the ecosystem attribute and the function it serves, in order to identify a threshold level of the perturbation beyond which the ecosystem function is impaired. In the habitat example (Sainsbury *et al.*, 1997) the ecological function of the benthos is to provide habitat complexity, in particular for the fish *Lethrinus* and *Lutjanus*, from which a threshold level might be obtained.

4.4 Summary and conclusion

Recovery is a rich concept that has no single definition in the context of the marine environment and ecosystems. WGECO use the following definition in the context of recovery capacity of ecosystem properties: a population or higher level ecosystem property is considered recovered if the necessary pieces for "normal" structure and function are present, even if some species historically observed are no more present or have modified abundance, biomass or age composition; this recovered state is not likely to be impaired by perturbations within the normal range of environmental variability and sustainable use, and can be attained and maintained without any special management measures.

This definition sets the scene for a rationale for positioning reference levels for ecosystem attributes with the objective of "sustainable use". A non-sustainable perturbation is one from which recovery is not likely to be rapid and secure. Thus, a reference point for sustainable perturbation might be derived by examining the relationship between perturbation level and recovery capacity. When this relationship is non-linear, a point where its slope changes most rapidly might provide an appropriate reference point, by analogy with the way reference points are selected on exploited stock spawning stock biomass (SSB) – recruit relationship (where SSB is an inverse measure of perturbation and recruit an index of stock recovery capacity); reference points are often selected where the slope of this relationship becomes steeper towards the origin. When this relationship is linear or non defined, or when ecosystem attributes have no capacity to recover from perturbation, then the same kind of analysis has to be performed with the ecosystem function served by this attribute instead of its recovery capacity.

Analyses of relationships between perturbation and ecosystem attributes should thus be undertaken at wide scale to help position reference points in a coherent and consistent manner across marine regions and indicators. Appropriate measures of ecosystem attributes are provided in Section 3.7. When trying to apply this framework to published case-history examples, WGECO found that functional relationships between recovery capacity and perturbation might be uncertain and only known qualitatively; or when it can be quantified the shape of the relationship might not suggest any obvious cutting point. Thus it is likely that analyses of the relationship between the perturbation and the ecosystem function served by various attributes will be required. However these are preliminary trials and further investigations are required to develop and apply this framework.

4.5 References

Aplet G.H., Keeton W.S. 1999. Application of historical range of variability concepts to biological Conservation. In: Practical approaches to the conservation of biological diversity (Baydasck R.K., Campa III,H., Haufler J.B., eds). Isaland Press, Washington DC.

Aronson, J.C., Le Floc'h, E., Ovalle, C., Pontanier, R. 1993. Restoration and rehabilitation of degraded ecosystems in arid and semiarid lands. II. Case studies in southern Tunisia, central Chile and northern Cameroon. Restor. Ecol. 3, 168–187.

ASCOBANS. 2009. Recovery Plan for Baltic Harbour Porpoises – Jastarnia Plan (Revision) (2009).

Borja, A., Elliott, M. 2007. What does 'good ecological potential' mean, within the European Water Framework Directive? Marine Pollution Bulletin 54, 1559–1564.

Brodziak, J.K.T., Cadrin, S.X., Legault, C.M., Murawski, S.A. 2004. Goals and strategies for rebuilding New England groundfish stocks ICES ASC 2004, Vigo, Spain, p. 32.

Caharer D.l., Henshaw J., Hall F., Knapp W.H., MacCammon B.P., Nesbitt J., Pederson R.J., Tiets C. 1992. Restoring ecosystems in the Blue Mountains: a report to the regional forester and the forest supervisors of the Blue Mountains forests. Portland, USDA Forest Service, Pacific Nort-west Region.

Cardoso, A.,C, Cochrane, S., Doerner H., Ferreira, J.G., Galgani, F., Hagebro, C., Hanke, G., Hoepffner, N., Keizer, P.D., Law, R., Olenon, S., Piet, G.J., Rice, J., Rogers, S.I., Swartenbroux, F, Tasker, M., and van de Bund, W. 2010. Marine Strategy Framework Directive, Management Group Report. Luxembourg Office for Official Publications of the European Communities. 59 pp.

CBD, 2008: UNEO/CBD/COP/9/29 Report of the Conference of the Parties to the Convention on Biological Diversity on the Work of its Ninth Meeting. Bonn, 2008.

DFO. 2002. Proceedings of the DFO Workshop on Implementing the Precautionary Approach in Assessments and Advice CSAS Proceedings 2002/009.

DFO 2005. A framework for Developing Science Advice on recovery targets for aquatic species in the context of the Species At Risk Act. DFO CSAS Proceedings 2005/057.

DFO. 2006c. A Harvest Strategy Compliant with the Precautionary Approach DFO CSAS Science Advisory report 2006/023.

DFO, 2007c. Revised Protocol for Conducting Recovery Potential Assessments. DFO CSAS Science Advisory Report 2007/039.

DFO. 2009. Stock Assessment of Northern (2J3KL) cod in 2009. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2009/009.

Elliot M., Burdon D., Hemingway K.L., Apitz SW.E., 2007. Estuarine, coastal and marine ecosystem restoration: confusing management and science – A revision of concepts. Estuarine, Coastal and Shelf Science, 74: 349–366.

FAO. 2008. Report of the Technical Consultation on International Guidelines for the Management of Deep-sea Fisheries in the High Seas. Rome, 4–8 February and 25–29 August 2008. FAO Fisheries and Aquaculture Report, No. 881. Rome, FAO. 2008, 42pp.

FAO. 2009a. FAO/UNEP expert meeting on impacts of destructive fishing practices, unsustainable fishing, and illegal, unreported and unregulated (IUU) fishing on marine biodiversity and habitats. Rome, 23–25 September 2009.

FAO. 2009b. International Guidelines for the Management of Deep-sea Fisheries in the High Seas. pp. 1–73.

Foden J, Rogers SI, Jones AP (2009) Recovery rates of UK seabed habitats after cessation of aggregate extraction. Mar Ecol Prog Ser 390:15–26

Fortibuoni T., Giovanardi O., Libralato S., Raicevich S., Solidoro C. 2008. Integrating historical naturalists' descriptions and statistical-scientific data to describe changes in fish assemblages over the last two centuries in the Northern Adriatic Sea (Mediterranean). ICES CM 2008/R:14.

Furness, R. W. and Greenwood, J. J. D. 1993. Birds as monitors of environmental change. Chapman and Hall, London. 368 pp.

Furness, R. W. and Camphuysen, C. J. 1997. Seabirds as monitors of the marine environment. ICES Journal of Marine Science, 54, 726–737.

Furness, R. W. and Tasker, M. L. 2000. Seabird-fishery interactions: quantifying the sensitivity of seabirds to reductions in sandeel abundance, and identification of key areas for sensitive seabirds in the North Sea. Marine Ecology-Progress Series, 202, 253–264.

Harris, M. P. and Wanless, S. 1990. Breeding success of British kittiwakes *Rissa tridactyla* in 1986–1988: evidence of changing conditions in the northern North Sea. Journal of Applied Ecology, 27: 172–187.

Harris, M. P. and Wanless, S. 1997. Breeding success, diet and brood neglect in the kittiwake (Rissa tridactyla) over an 11 year period. ICES Journal of Marine Science, 54, 615–623.

Hammer, C., Köster, F.W., St John, M., Hopkins, C.C.E., Wilson, D.C., Dorrien, C. von, and Strehlow, H. 2010. UNCOVER: The potential for success of recovery strategies for fish stocks/fisheries – Options and constraints. UNCOVER Synthesis Report No. 2 (in prep).

Hughes, T.P., Bellwood, D.R., Folke, C., Steneck, R.S., Wilson, J. 2005. New paradigms for supporting the resilience of marine ecosystems. TRENDS in Ecology and Evolution 20, 380–386.

Hutchings, J.A. 2000. Collapse and recovery of marine fishes. Nature 406, 882–885.

Lewis, S., Wanless, S., Wright, P. J., Harris, M. P., Bull, J. and Elston, D. A. 2001. Diet and breeding performance of black-legged kittiwakes *Rissa tridactyla* at a North Sea colony. Marine Ecology Progress Series, 221, 277–284.

Monaghan, P., Uttley, J. D., Burns, M. D., Thaine, C. and Blackwood, J. 1989. The relationship between food supply, reproductive effort, and breeding success in Arctic terns Sterna paradisaea. Journal of Animal Ecology, 58, 261–274.

ICES. 2001. Report of the Study Group on Further Developments of the Precautionary Approach to Fishery Management. ICES Document CM 2001/ACFM.

ICES. 2006. Report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO), 5–12 April 2006 , ICES Headquarters, Copenhagen. ACE:05. 174 pp.

ICES. 2008. Report of the working group on the Ecosystem Effects of Fishing Activities, ICES CM 2008/ACOM:41, Copenhagen.

ICES. 2009 Report of the working group on the Ecosystem Effects of Fishing Activities, ICES CM 2009/ACOM:20, Copenhagen.

Jackson J.B.C., Kirby M.X., Berger W.H., Bjorndal K.A., Botsford L.W., Bourque B.J., Bradbury R.H. *et al*. 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science, 293: 629–637.

Jennings, S., Dinmore, T.A., Duplisea, D.E., Warr, K.J., and Lancaster, J.E. 2001. Trawling disturbance can modify benthic production processes. J. Anim. Ecol. 70:459–475.

Levin PS, Fogarty MJ, Matlock GC, Ernst M. 2008. Integrated ecosystem assessments. National Oceanic and Atmospheric Administration. Available: http://www.nwfsc.noaa.gov/assets/25/6801_07302008_144647_IEA_TM92Final.pdf. Accessed 5 December 2008.

Lotze H.K. *et al*. 2006. Depletion, degradation, and recovery potential of estuaries and coastal seas. Science 312, 1806–1809.

Lotze H.K., Worm B. 2009. Historical baselines for large marine animals. Trends Ecol. Evol. 24(5), 254–262 (2009).

Manley P., Brogan G.E., Cook C., Flores M.e., Fullmer D.G., Husari S., Jimerson T.M., Lux L.M., McCain M.E., Rose J.A., Schmidt G., Schuyler J.C., Skinner M.J. 1995. Sustaining ecosystems: a conceptual framework. R5-EM-TTP-001. San Francisco, USA Forest Service, Pacific Southwest Region.

MarLIN Glossary. 2005. The Marine Life Information Network for Britain and Ireland: Online Glossary of Scientific Terms. Available from: <http://www.marlin.ac.uk/Glossaries/Gen_Glossary.htm>.

Mee, L.D., Jefferson, R.L., Laffoley, D.d.A., Elliott, M. 2008. How good is good? Human values and Europe's proposed Marine Strategy Directive. Marine Pollution Bulletin 56, 187–204.

Morgan P., Aplet G.H., Haufler J.B., Humphries H.C., Moore M.M., Wilson W.D. 1994. Historic range of variability: a useful tool for evaluating ecosystem change. Journal of sustainable forestry, 2 (1and2): 87–111.

Murawski, S. submitted. Rebuilding depleted fisheries: The good, the bad, and the mostly ugly. ICES Journal of Marine Science.

Pauly D. 1995. Anecdotes and the shifting baseline syndrome of fisheries. Trends Ecol. Evol. 10(10), 430.

Pinnegar J.K., Engelhard G.H. 2008. The 'shifting baseline' phenomenon: a global perspective. Reviews in Fish Biology and Fisheries, 18 (1): 1–16.

Portland USDA Forest service. 1994. Natural variability – Implications for Ecosystem management. Volume II: Ecosystem management principles and applications. In: Eastside forest ecosystem health assessment.

Rice, J. C. 2009. A generalization of the three-stage model for advice using the precautionary approach in fisheries, to apply broadly to ecosystem properties and pressures. – ICES Journal of Marine Science, 66: 433–444.

Rochet, M-J., Collie, J.S. Jennings, S., and Hall, S.H. Does selective fishing conserve community biodiversity? Predictions from a length-based multispecies model (In preparation).

Rosenberg A.A., Bolster W.J., Alexander K.E., Leavenworth W.B., Cooper A.B., McKenzie M.G. 2005. The history of ocean resources: modelling cod biomass using historical records. Front. Ecol. Environ. 3(2), 84–90.

Sàenz-Arroyo A., Roberts C.M., Cariño-Olvera M. 2005. Using fishers' anecdotes, naturalists' observations and grey literature to reassess marine species at risk: the case of the Gulf grouper in the Gulf of California, Mexico. Fish Fish. 6, 121–133.

Sainsbury, K.J., Campbell, R.A. Lindholm, R., and Whitelaw, A.W. 1997. Experimental management of an Australian multispecies fishery: examining the possibility of trawl-induced habitat modification. Amer. Fish. Soc. Symp. 20: 107–112.

Society for Ecological Restoration International Science and Policy Working Group. 2004. The SER International Primer on Ecological Restoration. www.ser.org and Tucson: Society for Ecological Restoration International.

Thom R. 1975. Structural Stability and Morphogenesis. Benjamin-Addison Wesley, New York. 1975.

Turnhout, E., Hisschemöller, M., Eijsackers, H. 2007. Ecological indicators: between the two fires of science and policy. Ecological Indicators 7, 215–228.

Wakeford, R.C., Agnew, D.J., and Mees, C.C. 2007. UNCOVER Deliverable 26: Updated review of existing recovery plans. Review of institutional arrangements and evaluation of factors associated with successful stock recovery plans. MRAG, March 2007. 58 pp.

Wakeford, R.C., Agnew, D.J., and Mees, C.C. 2009. Review of institutional arrangements and evaluation of factors associated with successful stock recovery plans. Reviews in Fisheries Science, 17(2): 190–222.

Worm, B., Hilborn R., Baum, J.K., Branch, T.A., Collie, J.S., Costello, C., Fogarty, M.J., Fulton, E.A., Hutchings, J.A., Jennings, S., Jensen, O.P., Lotze, H.K., Mace, P.M., McClanahan, T.R., Minto, C., Palumbi, S.R., Parma, A.M., Ricard, D., Rosenberg, A.A., Watson, R. and Zeller, D. 2009. Rebuilding global fisheries. Science. 325:578–585.

5 ToR c) Proportion of large fish EcoQO indicator

c) Continue to work on the proportion of large fish EcoQO indicator for the North Sea;

5.1 Introduction

The purpose of this Section is to bring together and summarise recent work on the development and operational application of the large fish indicator (LFI). WGFE has been tasked with carrying out some of this development work and key outputs from their endeavours are briefly reported here. Two manuscripts for publication in the ICES Journal of Marine Science (Greenstreet *et al*., in press; in review) have recently been produced and the new results from this work are briefly reviewed here.

The Ecological Quality Objective (EcoQO) for the North Sea's demersal fish community has been defined as *The proportion (by weight) of fish greater than 40 cm in length should be greater than 0.3* (Heslenfeld and Enserink, 2008) – this represents a management target. Managers will therefore need advice as to how this target might be achieved. WGECO has recognised the need for theoretical process-based models to provide the scientific basis for this advice. Specific scenarios have been suggested for these models to address in order to couch this advice in the most appropriate terms. WGFE has reviewed progress in the development a number of potentially useful models. Here we briefly summarise their conclusions and report on the more recent results using one of these models to address explicitly the scenarios proposed by WGECO.

The LFI was intended to quantify the status of the demersal fish community in the North Sea, which could then be compared to benchmark that would meet the vision of *healthy and biologically diverse oceans and seas*. Here we consider what information the metric does actually convey within this broader context.

The EcoQO process for the North Sea constituted a pilot study. OSPAR's original intention, if successful, was to roll out this approach across the remaining OSPAR regions. To some extent this has occurred; in producing the OSPAR Quality Status Report for 2010 (QSR2010) the LFI was applied to groundfish survey data collated for four of the five OSPAR regions. This raises the issue as to how the LFI should be tuned to best suit the different communities inhabiting each region. It raises issues regarding spatial variation in the metric and these are briefly examined here, not so much to provide solutions at this stage, but rather to raise the issue. The LFI has been adopted as one of the metrics to support the food webs descriptor of good environmental status (GES) in the European Union's Marine Strategy Framework Directive (MSFD). This again raises spatial issues as the metric will need to be applied in different marine areas covered by the MSFD.

5.2 Uptake of the "Large Fish Indicator"

The LFI has been developed to support the OSPAR EcoQO related to the 'Fish communities' issue, and ICES has for several years provided advice and science support on the indicator. In view of the advanced stage of development, the understanding of the LFI metrics already generated for regional seas, and links to management measures, the indicator has received attention from beyond the OSPAR community.

During the ICES/JRC process to identify indicators for the descriptors of good environmental status under the MSFD, the LFI has recently been suggested as one indicator of 'food webs'. The Commission decision (as required under Article 9(3) states, under the heading 'Structure of food webs (size and abundance)' a criteria 'Proportion of selected species at the top of food webs', as follows;

The rate of change in abundance of functionally important species will highlight important changes in food web structure. Indicators are to be developed for large fish (by weight) (4.5), using the experience in some sub-regions (e.g. North Sea). For large fish, data can be used from fish monitoring surveys, on an annual basis, at the scale of a regional or sub-regional sea.

The LFI has also been used in the UK as one indicator to describe 'marine ecosystem integrity' (http://www.jncc.gov.uk/page-4229), and also as a supporting indicator for the UK governments' Natural Environment Public Service Agreement, to monitor progress towards achieving the vision for *clean, healthy safe, productive and biologically diverse oceans and seas*.

As a result of this increasing level of interest, and the likelihood that other Member States may also become interested in the potential application of the LFI in their own waters (see Section 5.5), this chapter of the report briefly summarises the progress to date, and sets out guidance for how best to apply the LFI to the different marine regions covered by the MSFD (Figure 5.2.1).

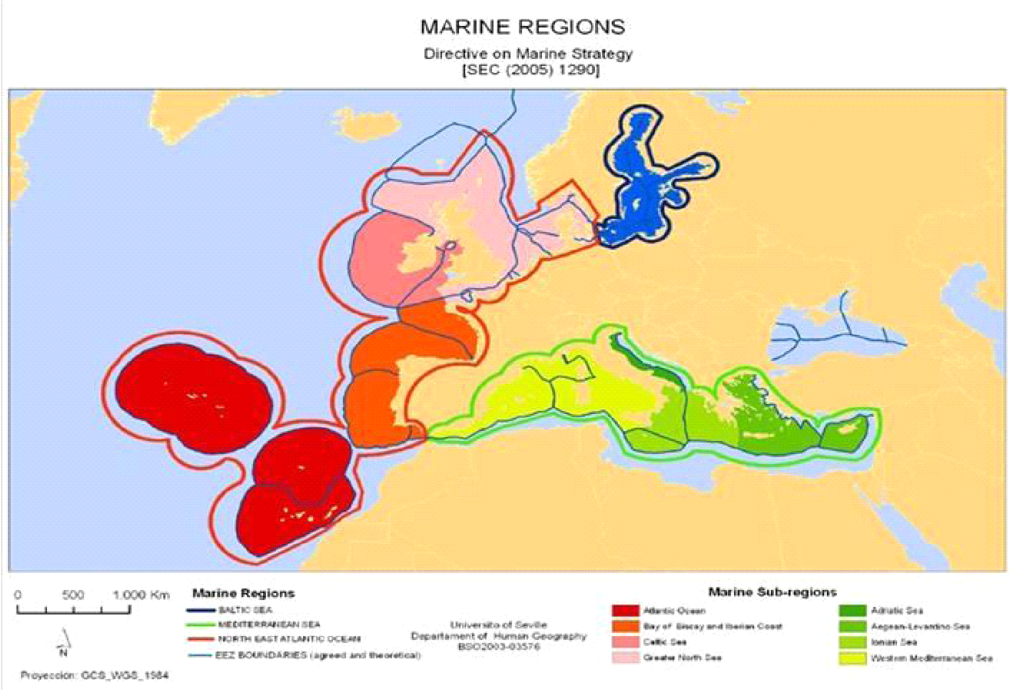


Figure 5.2.1. Marine regions covered by the Marine Strategy Framework Directive.

5.3 Recent developments in the North Sea "Large Fish Indicator"

Development of a LFI to support an EcoQO for the demersal fish community of the North Sea has been a long-term project of ICES. This history has recently been reviewed and documented (Greenstreet *et al.*, in press). Here we review the key results to have emerged from this latest analysis. The first quarter (Q1) International Bottom Trawl Survey (IBTS) data were reanalysed to update the LFI trend. The value of the LFI has continued to increase, standing at 0.22 in 2008 against an EcoQO target of >0.3. This represents a substantial improvement in the status of the North Sea's demersal fish community since its low point of 0.05 in 2002 (Figure 5.3.1).

Fishing was believed to be the most damaging anthropogenic activity affecting the North Sea's demersal fish community (Halpern *et al.*, 2008). The LFI was selected as the best indicator to support a North Sea demersal fish EcoQO because size-based metrics were considered most sensitive to the damaging effects of fisheries on demersal fish communities. Greenstreet *et al.* (in press) examined this relationship between the LFI and fishing mortality explicitly. Trends in both the LFI and community average fishing mortality in each year ($F_{com,y}$) are shown in Figure 5.3.2A. A lag of one year (the LFI in Q1 responds to F over the previous year) was considered as the baseline, but the relationship between the two variables was not significant (Figure 5.3.2B). Lags of between 12y and 18y, however, resulted in significant relationships between the LFI and $F_{com,y}$ (Figure 5.3.2C). Figure 5.3.2D shows the relationship with a lag of 16 years as an example, while Figure 5.3.2E shows the same two trends depicted in Figure 5.3.2A, but with 15 year lag introduced and the LFI trend inverted (the relationship was negative) allowing direct examination of the relationship. Inter-annual differences in both the LFI ($\Delta I_{LF,y}$) and community average fishing mortality ($\Delta F_{com,y}$) were calculated, where

$$\Delta I_{LF,y=Y} = I_{LF,y=Y} - I_{LF,y=Y-1} \text{ and } \Delta \overline{F}_{com,y=Y} = \overline{F}_{com,y=Y} - \overline{F}_{com,y=Y-1}$$

and $I_{LF,y=Y}$ (or $F_{com,y=Y}$) and $I_{LF,y=Y-1}$ (or $F_{com,y=Y-1}$) are the LFI (or community averaged mortality) values in any given year and in the previous year respectively, and the relationship between $\Delta I_{LF,y}$ and $\Delta F_{com,y}$ was examined. This was not significant at the baseline lag of one year, but a significant relationship was noted at a lag of two years. Changes in fishing mortality in either direction had an almost immediate effect in initiating predictable changes in the LFI.

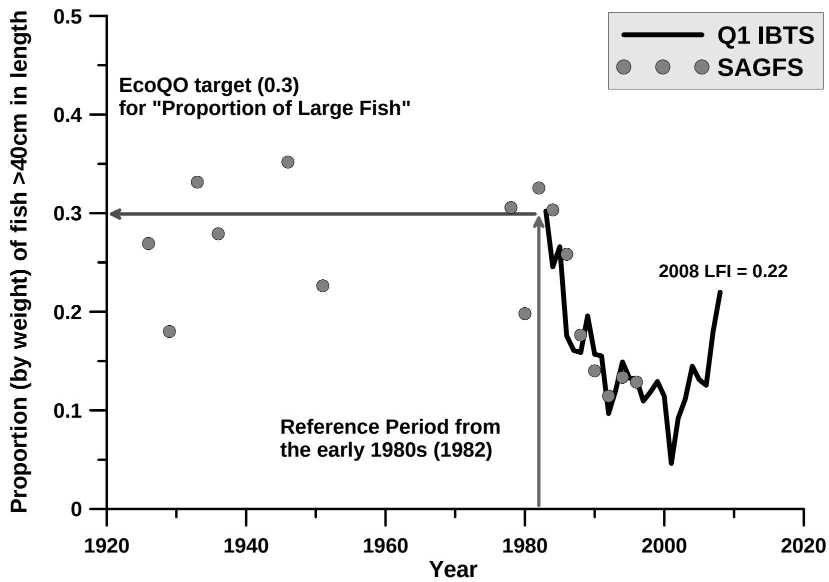


Figure 5.3.1. Variation in the redefined proportion of large fish indicator calculated for both the Q1 IBTS and the SAGFS data sets. The current LFI value is indicated, as is the EcoQO level for the indicator of >0.3 for the North Sea demersal fish community.

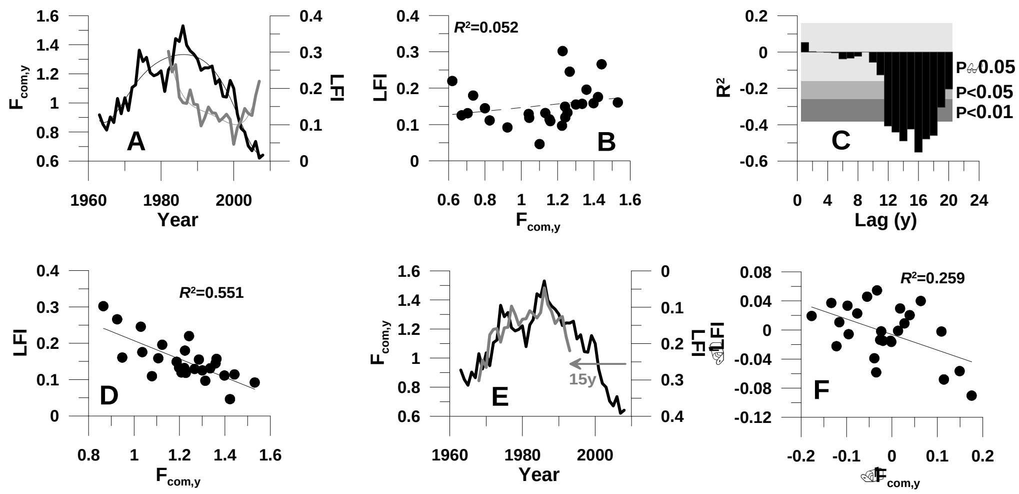


Figure 5.3.2. A: Temporal trends in community-averaged fishing mortality ($F_{com,y}$) and the LFI. B: Relationship between the LFI and community-averaged fishing mortality ($F_{com,y}$) with the baseline lag of 1y. C: Cross-correlation results showing R^2 values obtained in the correlation between the LFI and $F_{com,y}$ time-series with various lags built in (shading indicates significance levels). D: Relationship between the LFI and community-averaged fishing mortality ($F_{com,y}$) with a lag of 16y. E: Showing the effect of a negative lag of 15y on LFI time-series (shown inverted to take account of the negative correlation and make comparison easier) in comparison with the community-averaged fishing mortality ($F_{com,y}$) time-series. F: Relationship between the inter-anual differences in the LFI and community average fishing mortality.

Changes in fishing mortality in either direction had an almost immediate effect in initiating predictable changes in the LFI; a decrease in fishing mortality in one year over the previous year produced an increase in the LFI in January not of the following year, but in the year after that. The lag between the two time series themselves resulted from the numerous biological processes that follow changes in fishing mortality, most occurring in a pre-determined sequence. For instance, population age structure in large-bodied fish will continue to move towards a new equilibrium for many years following reductions in fishing mortality, and the same processes will operate in reverse among populations of small bodied fish as predation mortality rates rise with the increase in large fish abundance (Beverton and Holt, 1957; Daan *et al.*, 2005). Changes in fish community life history composition associated with varying disturbance levels (Jennings *et al.*, 1999; Greenstreet and Rogers, 2000) may further increase the time over which these processes take place.

For most of the period that Q1 IBTS data were available fishing mortality was declining (Figure 5.3.2A), the LFI trend therefore monitored changes in the demersal fish community during a phase of recovery. Fishing immediately removes large piscivorous fish but there is no equivalent immediate replacement of such fish following reductions in fishing mortality; it takes time for small fish to grow and become large. Furthermore, when top-down control on a trophic level is reduced by fishing down the large predators, bottom-up processes take over and competition within the trophic level increases as abundance rises (Carpenter *et al.*, 1987; Shurin and Seabloom, 2005). This may increase competition among small sized fish, possibly impairing growth rates of juveniles of larger bodied species (Lekve *et al.*, 2002). Time to equilibrium may well be longer during recovery than during the disturbance phase. In a recent meta-analysis, the greatest increase in the abundance of large fished species was observed in marine reserves that had been in place for longer than 15 years (Molloy *et al.*, 2009).

5.4 Review of developments in theoretical modelling

In this Section we review recent work carried out by WGFE and report the results of work undertaken subsequently using one theoretical model to examine explicitly the management scenarios posed by WGECO.

5.4.1 Work done by WGFE in 2009

The 2009 Report of the Working Group for Fish Ecology (ICES, 2009b, Chapter 2) presented work using a size-structured, multispecies model to simulate the past time-series of the LFI. This work is briefly summarised here.

5.4.1.1 The model

FishSUMS, the fish community model developed by Strathclyde University and Marine Scotland, is both length-structured and species-resolved. It has been developed to model past and future trends in the LFI as well as other size-based, biodiversity and life-history composition metrics. Key species are represented as length-structured populations. The current version has evolved from a cod-centric model used to test hypotheses regarding North Sea cod stock recovery taking account of interactions between cod and their main prey and predators. Species represented included cod, haddock, whiting, common dab, Norway pout, herring, sandeel, *Nephrops* and grey gurnard. Zooplankton, benthos and other small prey fish ecosystem components were simply modelled as size-spectra with each size-class modelled by chemostat dynamics subject to predation by size-structured species. Development of FishSUMS is documented in ICES (2008a; 2009a, b) and Guirey *et al*. (2008) and a full description is available in Speirs *et al*. (submitted).

The model was driven by fishing mortality (F) by species, year and length-class, derived mainly from ICES assessments (see ICES (2009b) for details) and has been hand-fit to a comprehensive North Sea data set: yearly total stock biomass (TSB) in Q1 from ICES assessments (ICES, 2007a); recruit numbers (ICES, 2007a); normalised population length distributions from the 1991 IBTS; stomach content data by length from the 1991 Year of the Stomach (Greenstreet, 1996) and grey gurnard stomach content data from Mackinson and Daskalov (2007); and yearly ICES assessed landings (ICES, 2007a). The model was run for an initial 100 year spin-up with zero fishing mortality, until equilibrium was reached for all species, followed by a subsequent 100 year run with F as in 1960. Thereafter, the model was run with the F time-series from 1960 to 2006. A good fit was achieved to all the data. Examples, showing the fit to TSB and 1991 length-distributions demonstrate that it was possible, with a relatively simple multispecies fish model, to simultaneously achieve a satisfactory fit to landings, recruits, stock biomass, population length distributions and diet data (Figures 5.4.1.1.1 and 5.4.1.1.2). At this stage, the model does not account for environmental variability or stochasticity, so it could not capture features such as the extreme variability in haddock recruitment.

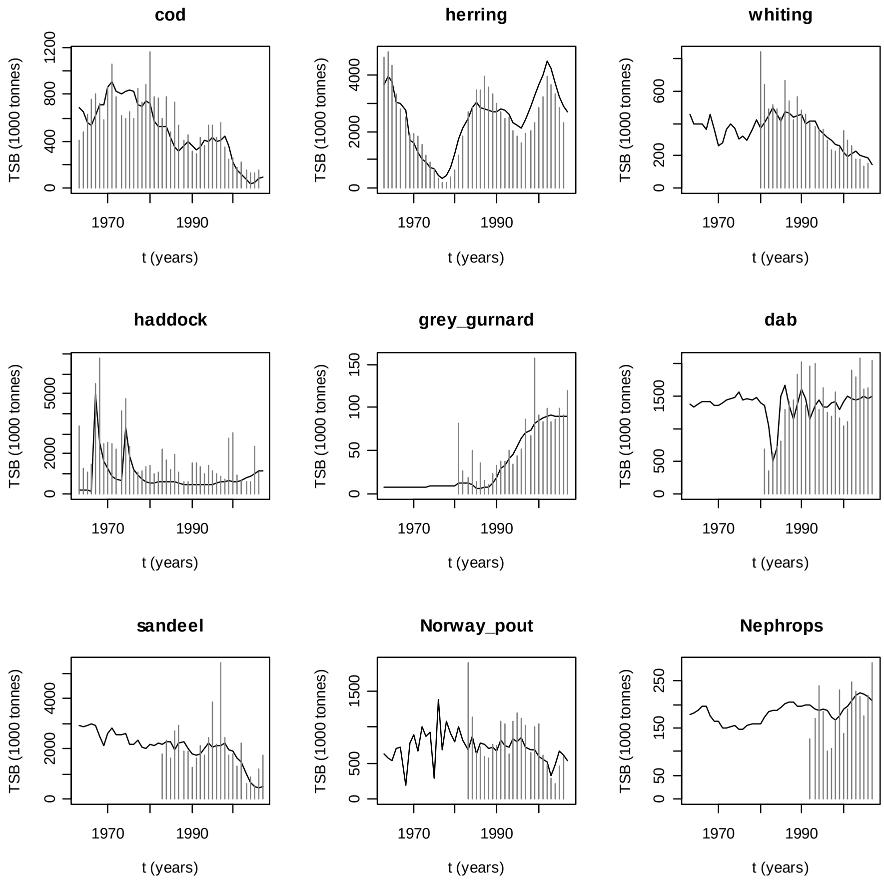


Figure 5.4.1.1.1. Time-series of modelled (solid line) and observed (grey bars) total stock biomass (TSB) of all the length-structured species included in the model.

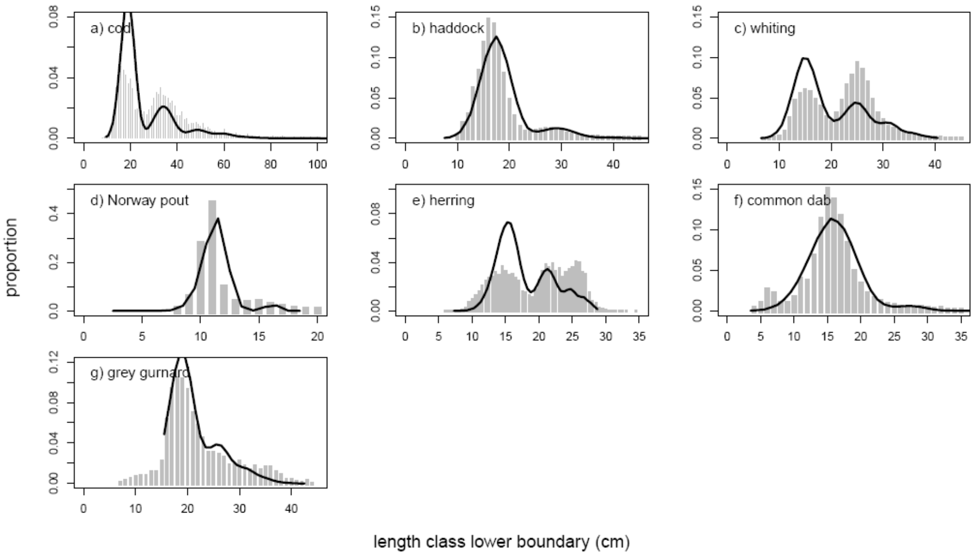


Figure 5.4.1.1.2. Modelled and observed normalised length distributions from the IBTS in the North Sea in quarter 1 of 1991. The y-axis shows proportion of the population in 1 cm length-classes (0.5 cm classes for herring).

5.4.1.2 Modelling variation in the LFI

The first task was to demonstrate that temporal variation in the LFI determined for the whole North Sea demersal fish assemblage could be adequately captured by modelling only a sub-set of species in this assemblage. Variation in the LFI determined for the sub-set of species currently included in the model, and for the sub-set ultimately intended to be included (ideal sub-set), was shown to be closely correlated with variation in the LFI determined for the full demersal species compliment (Figure 5.4.1.2.1). This confirmed that it was not necessary to model the full set of species caught in the IBTS in order to simulate behaviour of the LFI. "Partial ecosystem" models, typically incorporating ten to fifteen major species, can provide a basis for advice in support of an ecosystem approach to management to achieve EcoQOs for the North Sea's demersal fish community.

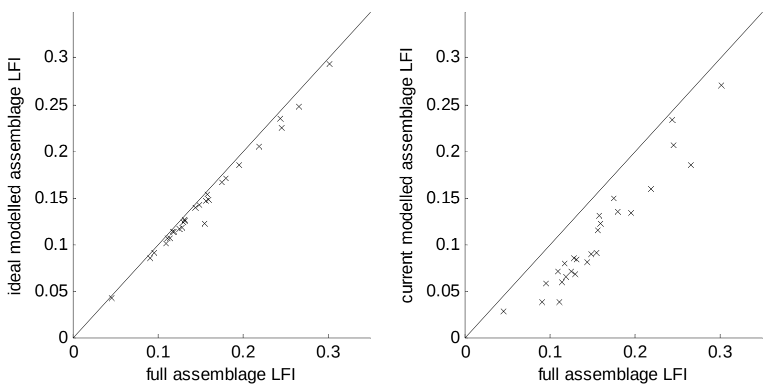


Figure 5.4.1.2.1. The LFI calculated from Q1 IBTS data from 1982 to 2008 for (1) all demersal species caught in the survey, (2) an assemblage comprising the 15 species making up the top 95% of the total demersal biomass (the "ideal modelling subset") and (3) the set of species currently represented in the model. The bottom panel compares the three data sets. Solid lines in these plots indicate the line with slope 1.

5.4.1.3 Simulating past trends in the LFI

The FishSUMS model was adapted to include a routine for calculating the LFI in Q1 for all years from 1982. The IBTS survey was emulated by applying length- and species-dependent trawl selectivity coefficients to the numbers output by the model. These empirically-derived selectivities were taken from the analysis by Fraser *et al.* (2007) comparing MSVPA output with IBTS data for the years 1998 to 2004 (see Figure 5.4.1.3.1). The LFI was then calculated by summing the selectivity-corrected biomass of demersal species above and below the 40 cm threshold (Figure 5.4.1.3.2). The modelled LFI trend was compared with the actual IBTS-derived LFI trend based on the whole demersal fish assemblage and on just the subset of species represented in the model (Figure 5.4.1.3.2 top and bottom panels respectively). Both pairs of trends were highly correlated, with correlation coefficients of R^2=0.69 and R^2=0.78 ($p < 0.001$ for both, 2 d.f.) for the full and modelled assemblage respectively.

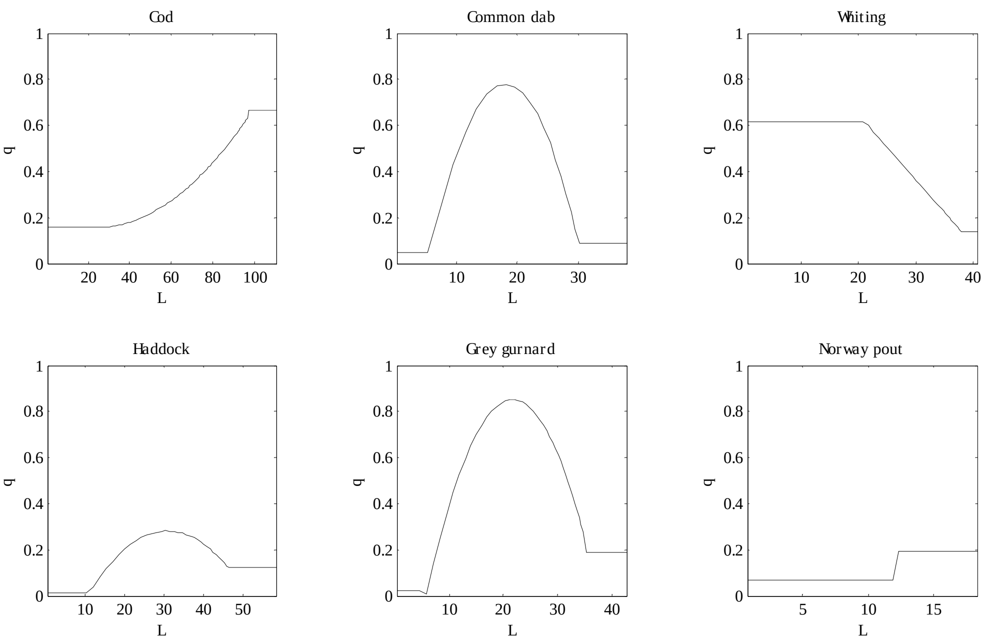


Figure 5.4.1.3.1. Selectivity coefficients at length from Fraser *et al.* (2007).

While the model may have captured trends in the LFI well, modelled LFI values were generally much lower than actual observed values. This may partly be explained by the fact that saithe are currently not included in the modelled suite of demersal species, and this is a relatively abundant species that grows to large body size and which would therefore be expected to contribute considerably to the biomass of fish >40cm in length. Intentions are therefore to build this species into the model as it is developed further in the future. However, the fact that the model output fails to closely match actual LFI values whilst closely matching TSB and length distributions suggested that the main problem lay with the way that the IBTS was emulated. If assumed selectivities in the survey gear are incorrect, the LFI time-series will be "biased" despite the good fits to the ICES assessment data. Fraser *et al.* (2007) discuss various issues that may impact on the way that their coefficients might be used to simulate the IBTS survey. For example, the study does not assess spatial or inter-year variability in catchability and can not account for the full length range of each fish species. Furthermore, and perhaps most critically, their catchabilities are based on Q3 IBTS data, whilst the LFI is calculated using Q1 data.

5.4.2 Predicting LFI behaviour under future fishing scenarios

Although issues still remain to be resolved with the FishSUMS model, selectivity of the IBTS survey requires "tuning", key demersal species such as plaice, starry ray, monkfish and saithe still require inclusion, the model nevertheless captures well the underlying historic trend in the LFI. The model has therefore been used to investigate the behaviour of the LFI under different future fishing scenarios. Because of these issues still requiring resolution, however, relative comparisons should be made rather than absolute comparisons. The EcoQO target of >0.3 was selected because this was the value of the LFI in the early 1980s when demersal fishing in the North Sea was last deemed to be sustainable. Thus, rather than judging the performance of the modelled LFI against a strict target of >0.3, it should instead be judged against the modelled value prevailing around 1983, i.e., a value of >0.08.

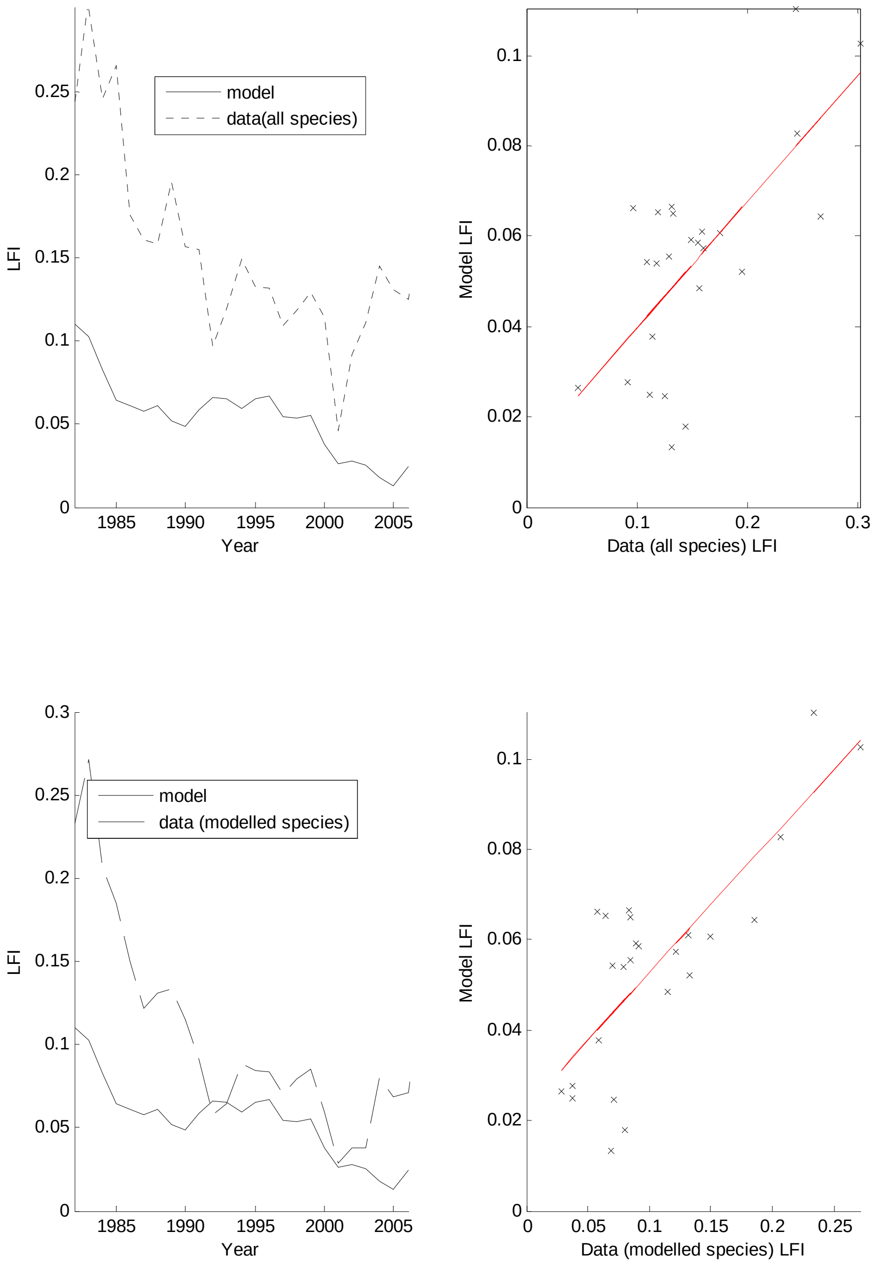


Figure 5.4.1.3.2. Time-series of the LFI calculated from the IBTS and model output (left plots). The data LFI is calculated from the whole IBTS-sampled assemblage (top) and from the modelled assemblage (bottom). The right panels show the data and model LFI in each year plotted against one another with the line of best fit in red.

5.4.2.1 Future fishing scenarios

Three different scenarios were examined. The simulations undertaken address the questions posed by WGECO (Table 5.2.1, ICES, 2008b).

5.4.2.1.1 Scaling of *status quo* fishing patterns

The first scenario was to continue fishing with the same fishing mortality, with respect to species and length, as in 2006, but to scale this fishing mortality ($F_{2006,s,l}$) by a constant factor (k), which was varied between 0 and 2 for all years after 2006 up to 2050, for all species and for all length classes.

$$F_{y,s,l} = kF_{2006,s,l}$$

Thus, a scaling of k=0 corresponds represent zero fishing mortality between 2006 and 2050; k=1 means continuing to fish exactly as in 2006 every year afterwards up to 2050 (*status quo*); k=2 represents a doubling of 2006 levels of fishing mortality between 2006 and 2050; k=0.5 represents a halving of 2006 levels of fishing mortality between 2006 and 2050. Because the fishing mortality by species at length is scaled by a constant factor, the distribution of fishing mortality between the different species and lengths remains the same as it was in 2006.

5.4.2.1.2 Fishing at F_{pa}

The second scenario was to fish all assessed demersal species at F_{pa} levels from 2006 until 2050. Again, the fishing mortalities were set by simply scaling 2006 levels of fishing mortality by species and length class ($F_{2006,s,l}$) by average fishing mortality across the fished age classes in 2006 ($F_{\overline{2006}}$) by F_{pa} for each species ($F_{pa,s}$). Thus, the scaling factor $F_{pa,s} / F_{\overline{2006}}$ equates to fishing mortality at F_{pa} for all species and length classes from 2006 onwards.

$$F_{y,s,l} = \frac{F_{pa,s}}{F_{\overline{2006}}} F_{2006,s,l}$$

This equation was applied to the assessed commercial species included in the model. Fishing mortality on non-assessed pelagic species was maintained at 2006 levels. This scenario is aimed at addressing the question of whether fishing at F_{pa} would be sufficient to bring the LFI up to the target value. In other words, do managers need to do anything more than single-species management at F_{pa} to achieve the broader community objective?

5.4.2.1.3 Cessation of fishing for some species

The third scenario repeated the one above except that fishing mortality on one fish species at a time was set to zero. Four assessed demersal species were included in the model, so four runs were carried out. For each run, F was set to 0 for all years after 2006 for the selected species. The remaining species were fished following the process described in Section 5.4.2.1.2.

5.4.2.2 Results

Figure 5.4.2.2.1 illustrates the results of scaling current day fishing by a constant factor. The simulations take around 20 years to come to equilibrium, with some transient oscillation. F scaled by anything less than 1.2 times 2006s levels leads to an increase in the LFI from the current value. However, unless scaling factors less than or equal to 0.8 are applied, the equilibrium LFI fails to reach a level equivalent to the modelled 1983 value, implying failure to achieve the EcoQO. Using a scaling factor of 0.8 produces an equilibrium LFI equivalent to the model 1983 value by 2020. This suggests that fishing mortality has to be reduced to 80% of 2006 levels if the EcoQO for the demersal fish assemblage is to be achieved by the date set for reaching good environmental status in the Marine Strategy Framework Directive. A scaling factor of 0.2

leads to the modelled LFI reaching levels equivalent to the 1983 value by 2010 but goes on to produce an equilibrium LFI of twice the 1983 level by 2020. F scaled by values greater than 1.4 times the 2006 level lead to decreases in the LFI from its current value.

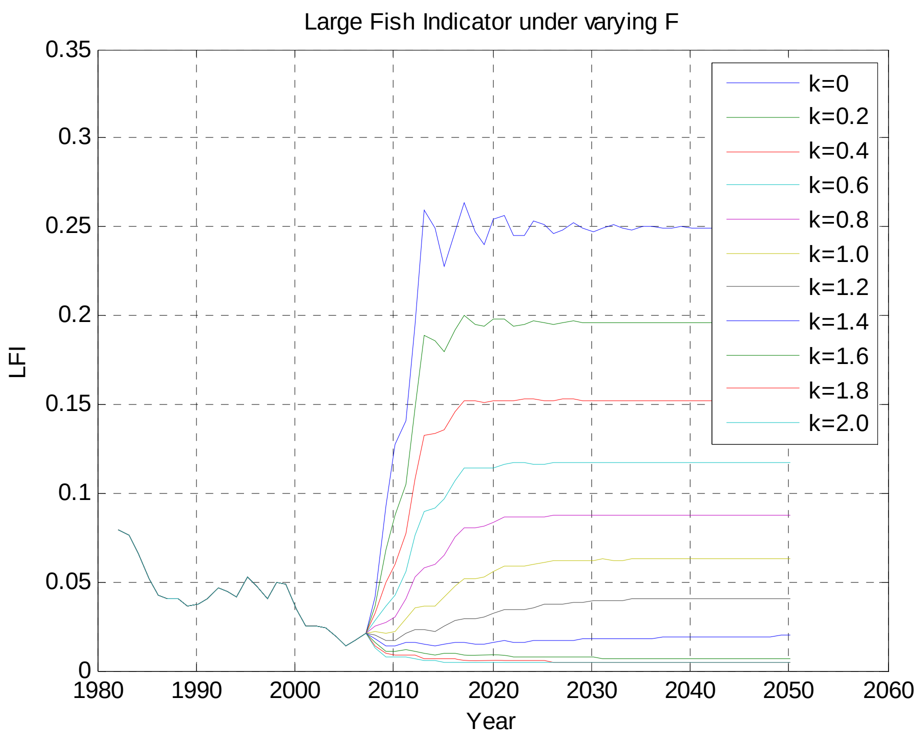


Figure 5.4.2.2.1. LFI under fishing mortality scaled by 2006 levels. F(year, species, length)=k*F(2006, species, length) for all years after 2006. See text for details.

Figure 5.4.2.2.2 shows results of scaling current day fishing by F_{pa} for the demersal assessed species. Again, the simulation takes around 20 years to fully equilibrate. The equilibrium LFI value is almost exactly at the modelled 1983 level. This result suggests that simply managing fisheries in line with the principles underpinning the current precautionary approach would be sufficient to ensure that the EcoQO for the demersal fish assemblage would be met by 2020.

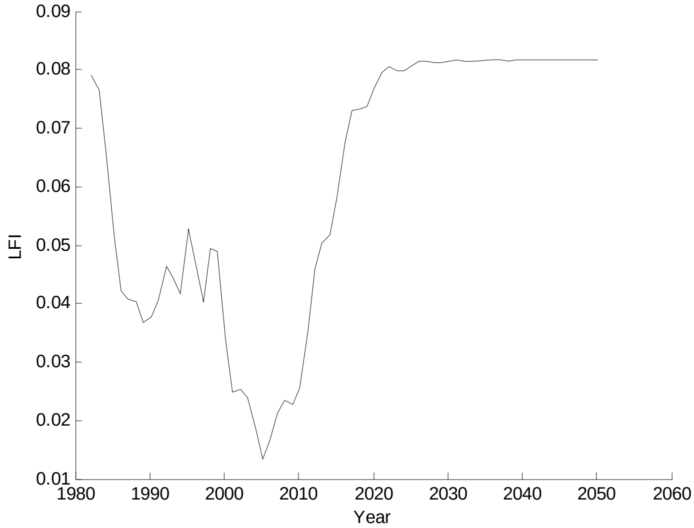


Figure 5.4.2.2.2. LFI under Fpa fishing mortalities. F(2006,species,length) is scaled by Fpa/Fbar for each assessed demersal species. For other species, F is as for Figure 6. See text for details.

Figure 5.4.2.2.3 shows the results of setting F to 0 for each assessed demersal species in turn, with mortality set at F_{pa} for the remaining species. These runs clearly show the importance of cod in contributing to the biomass of fish over 40 cm. The simulations setting haddock, Norway pout or whiting F to 0 are similar to the above run with all species at F_{pa}. In this case, the steady state LFI is at about the 1983 levels. The simulation with cod F at 0 is very different, with a rapid increase in the LFI to three times the 1980 level and equilibration by 2015.

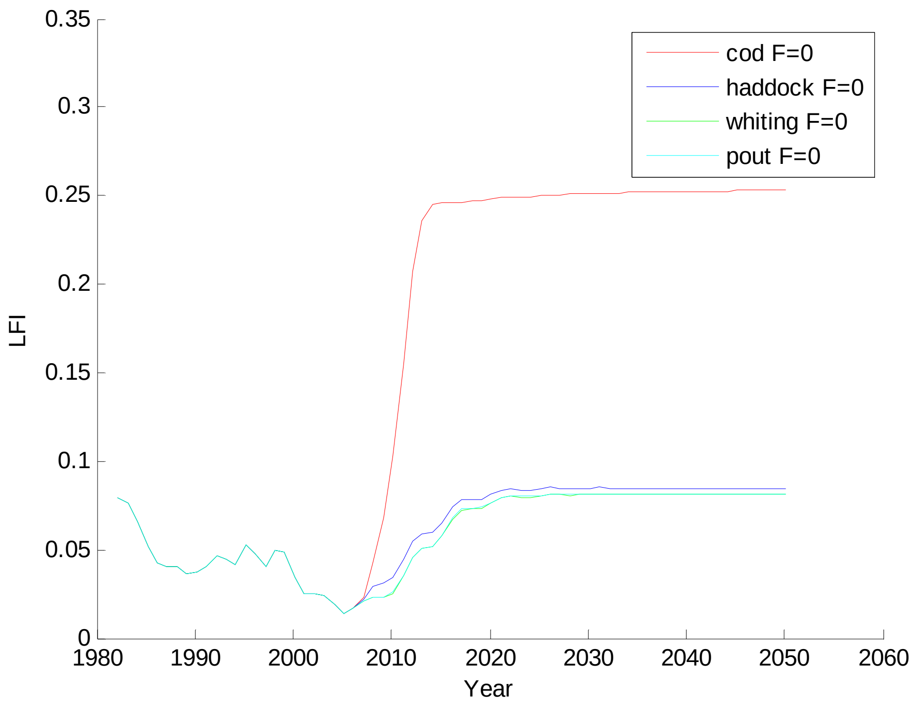


Figure 5.4.2.2.3. LFI under Fpa fishing mortalities with each assessed demersal species F set to 0 in turn. F(2006,species,length) is scaled by Fpa/Fbar for each assessed demersal species. For other species, F is as for Figure 6. See text for details.

5.4.2.3 Conclusions

- A multispecies model has been developed that is capable of fitting simultaneously to several datasets. It is size-structured, fit to the North Sea, capable of representing the population dynamics of individual species, and driven by fishing mortality. It is therefore suited to addressing questions about size-based community metrics (ICES, 2008b).
- It has been shown (ICES, 2009b), that a model need not represent the full North Sea demersal assemblage to simulate the LFI. With the current suite of species included, the model is able to reproduce past trends in the LFI.
- Difficulties currently exist in emulating the IBTS and, therefore, in reproducing quantitatively the exact LFI historic time-series, although in relative terms the trend is reproduced.
- The model should be improved by extending to the latest assessment year (2009) and adding further key demersal species such as saithe and plaice.
- If the model LFI from the 1980s is used as the reference level, the model runs suggest that fishing at 0.8 times present day levels, or at Fpa for each demersal species and at present day levels for the other species, is predicted to achieve the EcoQO by 2020.
- A scenario of constant fishing mortality is perhaps unrealistic. More realistic fishing scenarios in which managers respond to transient peaks and troughs in the abundance of large fish could be explored. This is important in the context of the 10–20 year period of transient oscillation seen in all runs.

- A consistent feature in all model runs was the long time required for the FFI to reach equilibrium. At F_{pa}, the time to equilibrium period was of the order of 14 years.
- The model runs do not take account of environmental/recruitment variability and therefore cannot adequately capture the variability in "small fish" biomass.
- The model runs suggest that, within the current model set up, the LFI is largely driven by cod.

The model output should be compared with similar simulations using different models, e.g. the IMAGE multispecies size-spectrum model presented in ICES (2008b) or pure size-based models. Such comparisons will ensure that management advice is based on a broad scientific knowledge base, and will aid the identification of key processes involved in regulating the structure and composition of the demersal fish assemblage.

5.5 Sub-regional spatial variation within the North Sea

The EcoQO for the North Sea demersal fish community has been set at a regional scale; a single target of >0.3 has been established for the entire North Sea. However, the demersal fish community in the North Sea is not homogeneous; the structure and composition of the community varies in space and the pattern of variation is consistent over time (Daan *et al*., 1990; Fraser *et al*., 2008). Examination of spatial variation in the LFI across the North Sea revealed distinct patterns that were essentially consistent over time, as well as consistent with spatial variation in the structure and composition of the community determined using other univariate community descriptors (Fraser *et al*., 2008; Ehrich *et al*., 2009). LFI values were consistently lowest in a large area of the northwestern North Sea (Figure 5.5.1).

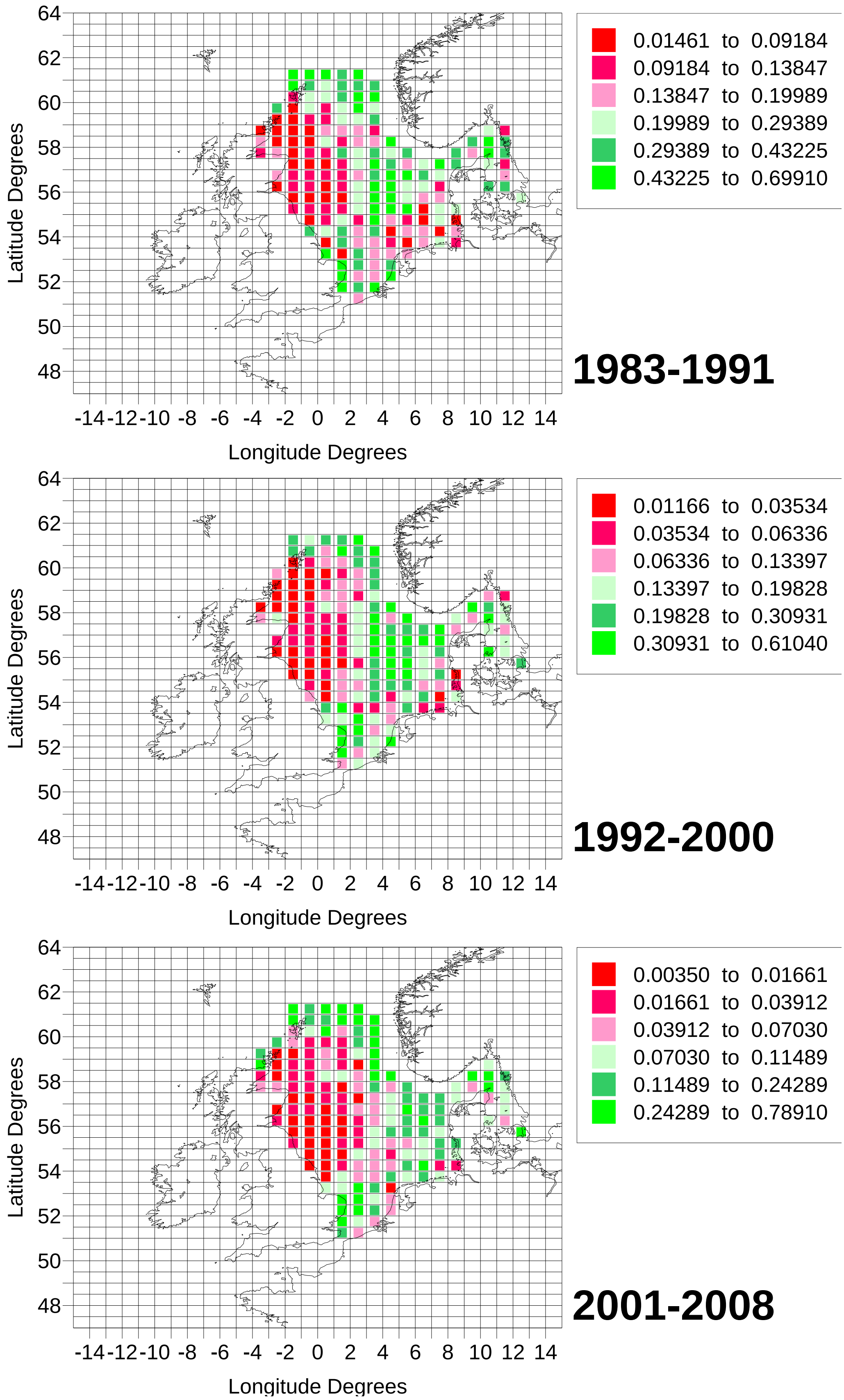


Figure 5.5.1. Spatial variation in the LFI across the North Sea in three separate time periods. Note that the same colours refer to different values in different time periods.

Two questions therefore arise:

- What would be the consequences of implementing an ecosystem approach to management for the wider demersal fish community using a single EcoQO for the the North Sea, compared to using separate EcoQOs be set at a sub-regional scale?
- Can understanding variation in LFI trends at sub-regional scale, combined with data on sub-regional variation in pressure regimes, assist in achieving management targets set at a regional scale?

At present we are not in a position to answer these questions, principally because of a lack of sufficient information regarding differences in levels and trends in fishing pressure in different parts of the North Sea. However, increasing access to vessel monitoring by satellite (VMS) data (Stelzenműller *et al.*, 2008; Lee *et al.*, 2010), combined with more detailed analysis of official landings and effort statistics (Greenstreet *et al.*, 2009), should allow this shortcoming to be addressed. In the meantime, analyses of LFI trends in different sub-regions of the North Sea were undertaken as a preliminary scoping exercise.

The area of the North Sea covered by the Q1 IBTS survey was sub-divided into eight sub-regions (Figure 5.5.2). These were the sub-divisions used in analyses undertaken as part of the assessment of the status of fish communities across the OSPAR regions for the 2010 Quality Status Report (Greenstreet *et al.*, 2009). Considerable variability between LFI trends in each sub-region was observed (Figure 5.5.3). In all sub-regions, initial declines in the early 1980s occurred. Evidence of recovery was apparent in some sub-regions, but not in others. The analysis identified one area, the western-central basin, where the LFI declined steadily between 1983 and 2000, from 0.16 to 0.02, with no evidence of any subsequent recovery. Similarly the LFI in the eastern-central basin currently stands at the lowest levels recorded in the area following an earlier recovery phase during the 1990s. Understanding the situations giving rise to the trends in these two areas, and addressing the problems identified, could contribute considerably to attaining a regional LFI EcoQO of >0.3.

Finally, actual quantitative values of the LFI varied considerably between different sub-regions, for example the LFI varied between 0.02 and 0.16 in the western-central basin compared with a range of between 0.2 and 0.7 in the Norwegian Deeps sub-region. This implies that either different EcoQOs would need to be set for each sub-region, or the definition of the LFI used in each sub-region might be “tuned” to more suit the metric to the varying nature of the fish community resident in each sub-region, and so make the LFIs more equitable quantitatively across regions.

Understanding sub-regional spatial variation in the behaviour of the LFI in a data-rich region such as the North Sea, where the processes governing the structure and composition of the demersal fish community are so well known, will greatly aid the roll out and application of the LFI to data from other marine regions.

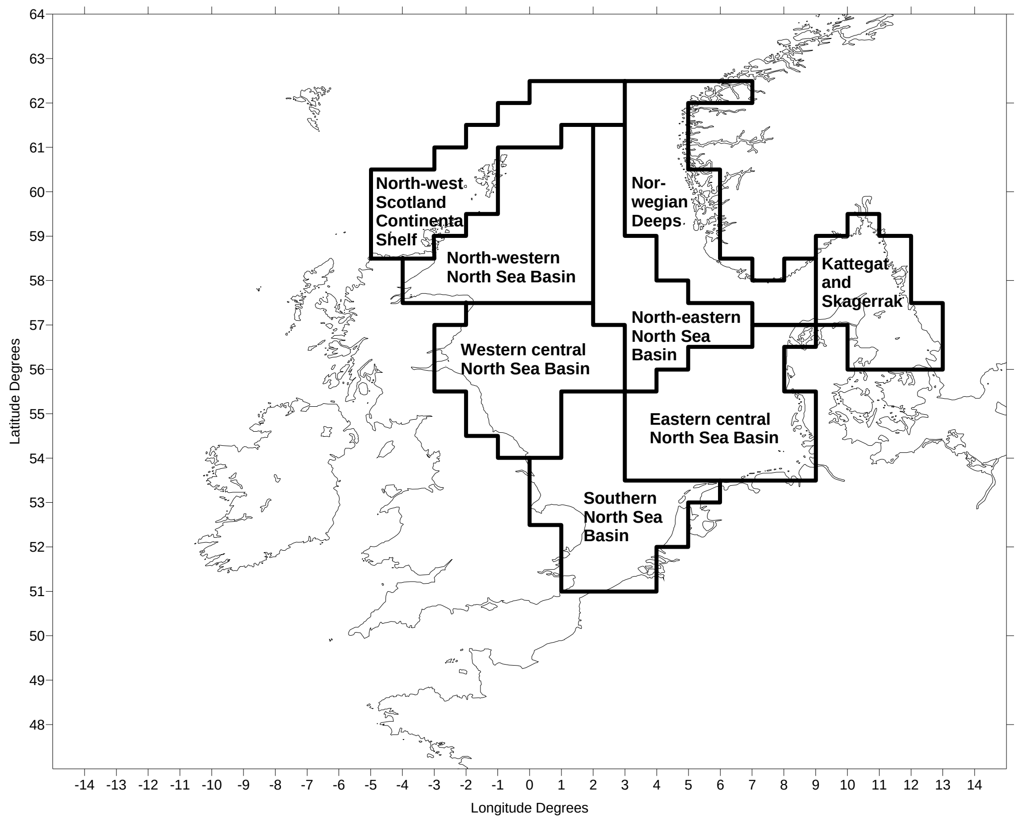


Figure 5.5.2. Chart of the North Sea showing boundaries of eight sub-region divisions. The area included in these sub-regions is the area covered by the IBTS Q1 survey. It covers most of OSPAR region II, but is not the whole of the region. OSPAR region II includes the Channel for example.

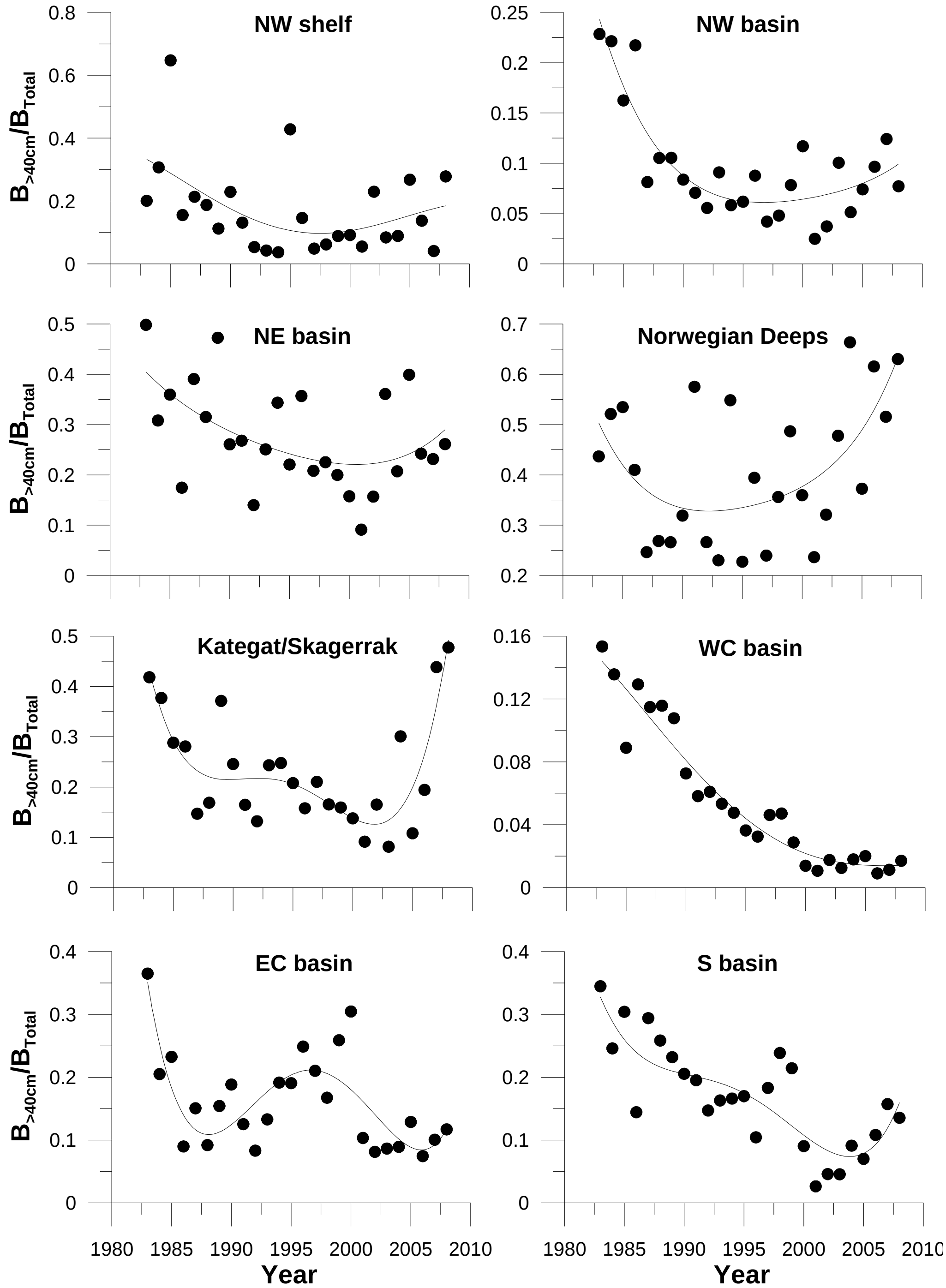


Figure 5.5.3. Variation in the LFI calculated for 8 sub-areas within the North Sea for which Q1 IBTS data were available. See Figure 5.5.2 for area key.

5.6 Variation in the LFI in different marine regions

In this Section we report on recent analyses of trends in LFIs calculated using various different ground fish data sets in marine regions other than the North Sea.

5.6.1 Trends in the Celtic Sea LFI

5.6.1.1 Introduction and methods

An investigation was made into the application of the LFI to an exploited marine system outside the North Sea. The Celtic Sea was chosen as it represents a heavily-fished region having a differing fish community and oceanography to the North Sea and is described by shorter and less spatially intensive survey time-series. The study was broadly intended to determine whether the LFI could be applied outside the North Sea in a direct and methodologically prescriptive manner, or whether system-specific interpretation or tuning would be required to produce a useful metric (where tuning refers to the process undertaken by WGFE (ICES, 2007b) to determine at what length fish are defined as "large" and whether the metric be computed on the basis of abundance-at-length or weight-at-length). A key constraint was that the UK West Coast Groundfish Survey (WCGFS) – the most enduring fisheries survey in the Celtic Sea – concluded in 2004. French and Irish Groundfish Surveys (IGFS) are ongoing but have somewhat different survey stations and seasonal coverage.

A series of objectives addressed the need to produce an informative LFI time-series:

1) Calculate a Celtic Sea LFI series extending from the first available survey year (1984-WCGFS) until the most recent completed survey (2009-IGFS).

2) Determine whether LFI values for the overlapping years (1997–2004) were sufficiently similar between survey-series that current values for the IGFS could be robustly compared to historical values of the WCGFS.

3) Define a target reference point (EcoQO) for the Celtic Sea based on values of the LFI for some period before major fisheries exploitation (the North Sea EcoQO uses the early 1980s as a reference period).

Given the larger size profile of the Celtic Sea fish community compared to the North Sea, three values of the large fish size threshold (40 cm in the North Sea) were tested (40, 42 and 45 cm). Additionally, separate LFI trends were determined for both the eastern and western Celtic Sea, and for the combined area, in order to evaluate possible spatial differences in community composition.

A further broad objective expressed the need to understand the ecological basis and drivers underlying trends in the Celtic Sea LFI:

4) Assess components and processes in the fish community that drive shifts in the LFI. This understanding is critical and allows robust inference about community state and the ecological meaning of shifts in such a univariate indicator. In the North Sea, the LFI is thought to be primarily influenced by an indirect effect of fishing - enhanced numbers of small fish following predation release that occurs when large piscivores are removed by fishing.

Trends in biomass (kg/km swept area^2) of the large and small components of the Celtic Sea fish community were investigated at species level. Shifts in these components (due to growth or immigration) and the response of the indicator to such shifts were studied to yield understanding of mechanisms driving the indicator.

5.6.1.2 Results and conclusions

A combined LFI series was defined, extending from 1986–2009. A large fish threshold of 40 cm was retained since changing this value had little effect on the LFI (Figure 5.6.1.2.1.). Given minimal differences in the LFI between eastern and western regions, a single spatial Celtic Sea indicator series was used (Figure 5.6.1.2.1.). While values of the LFI for overlapping years expressed a seasonal effect, they were within the same range (Figure 5.6.1.2.2). Based on a sequence of values for the period 1986–1990, a provisional EcoQO for the Celtic Sea LFI of 0.6 was defined. This may change if the species suite included in the index is further modified (see below). A sustained decline in biomass of large fish, compared to the absence of a clear trend in biomass of small fish suggested that the Celtic Sea LFI was being driven primarily by the direct removal of large fish – the direct effect of fishing. However, a key emergent issue was definition of the fish community complex from which the LFI should be calculated. In the North Sea this complex includes all species considered to be sampled in a representative way by the demersal GOV trawl and is therefore by default exclusively demersal. For the Celtic Sea series the inclusion of intermittently large catches of boarfish, sometimes defined as a demersal species, dominated the LFI in certain years and hence this species was ultimately excluded on the basis that it often forms large and poorly sampled pelagic shoals. Conversely, there is evidence that blue whiting, often considered a pelagic species, has a demersal juvenile phase where it can represent a critical prey resource for piscivorous species in the Celtic Sea and may be adequately sampled by the GOV. Hence the indicator will now be re-calculated to include this community component. Importantly, moderate to large catches of this species during the reference period will likely push the LFI down for these years resulting in a lower EcoQO value. It may also shift the indicator from being driven by large fish to small fish.

Extending the LFI concept to other marine regions will likely require system-specific interpretation based on a sound theoretical framework and careful investigation. The conclusions of the North Sea LFI would be more useful expressed as a flexible protocol than a strictly prescriptive mechanism.

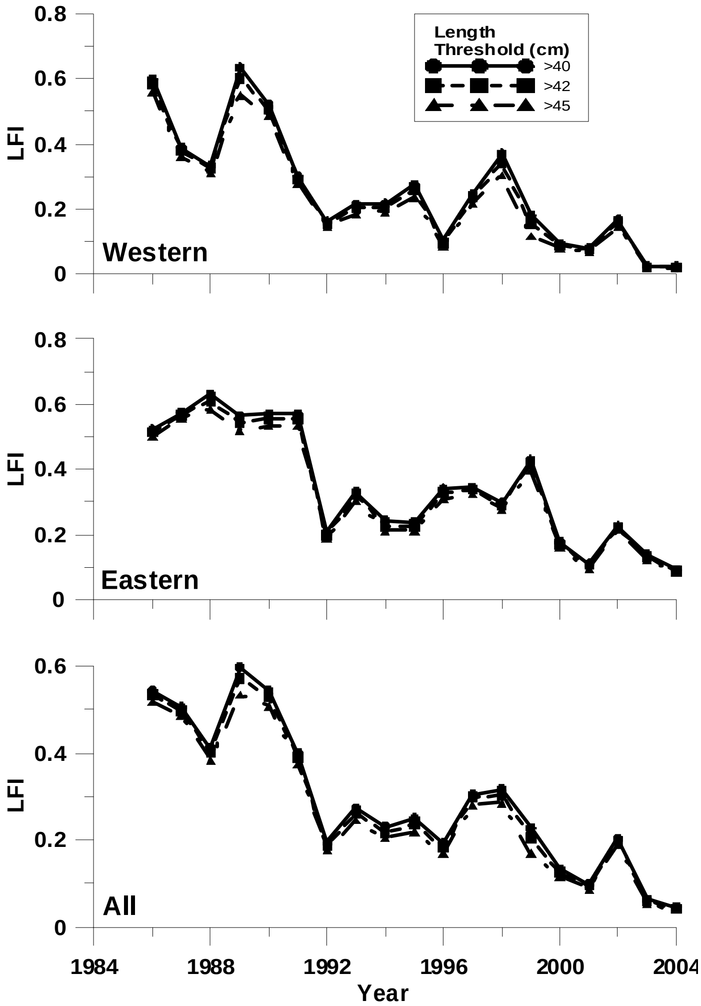


Figure 5.6.1.2.1. Time-series of the Large Fish Index (LFI) for each of the western and eastern Celtic Sea and for a combined series. The three lines on each plot represent the LFI calculated using a 40 cm, 42 cm and 45 cm threshold for 'large' fish.

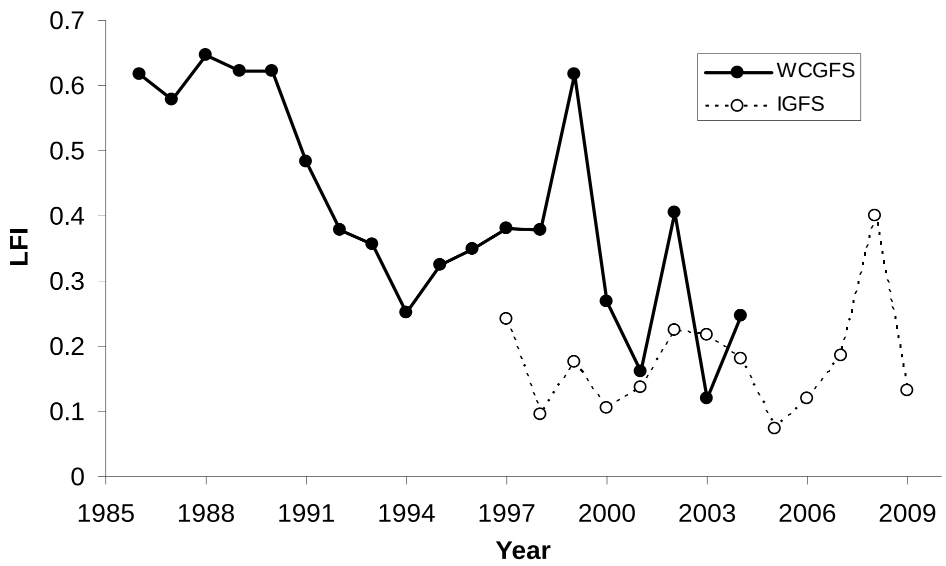


Figure 5.6.1.2.2. A time-series of the large fish indicator based on the UK West Coast Groundfish Survey (WCGFS) and the Irish Groundfish Survey (IGFS) in the Celtic Sea.

5.6.2 South Western Waters RAC-Bay of Biscay, Atlantic Iberia and the Azores Archaepelago

The LFI has been calculated for the South Western Waters (SWW) RAC region (Borges *et al.*, 2010). The method applied followed procedures used in the North Sea.

5.6.2.1 Methods

The LFI was determined using data obtained from four different fishery independent surveys, reporting cpue of species by length, undertaken in four marine sub-regions within the SWW RAC. In the Azorean region the LFI was also calculated using the Azorean deep longline (DLL) survey data because suitable bottom trawl survey were not available (Table 5.6.2.1.1). Regional weight-at-length relationships were used to convert abundance-at-length to weight-at-length. Development of the LFI involved sensitivity analysis of the metric to reduce noise from recruitment events and enhance its sensitivity to the effects of fishing on the fish community associated with the removal of large fish. Different length thresholds were therefore used to define "large" fish; 20 cm, 30 cm, 40 cm. Longline gears are selective for larger specimens so for the Azores sub-region, larger threshold limits of 50 cm and 60 cm were also used. In the North Sea the metric relates only to the demersal fish community; species considered to be pelagic are excluded from the analysis. The consequences of a similar exclusion of pelagic species on the LFI calculated for the SWW RAC region was examined by calculating the metric based on all species sampled and with species considered to be pelagic excluded.

Table 5.6.2.1.1. Surveys and data used for determining the LFI in the SWW RAC. In brackets years not considered given a change in gear or vessel.

Survey	Type Survey	Gear	Data Series(*)	No. Years(*)	No. Species
EVHOE BoB(**)	Bottom trawl	GOV 36/47	1997–2007	11	190
SPNGFS	Bottom trawl	Baka 44/60	1992–2007	16	180
PGFS	Bottom trawl	Campelen 1800/96	1989–2008 (96, 99, 03-04)	20 (4)	199
Azores DLL	Demersal long-line	Longline	1996–2008 (98, 06)	13 (2)	113

5.6.2.1.1 Individual survey details

Bay of Biscay, survey EVHOE BoB area

The French demersal survey began in 1987, but there was a change in vessel and sampling design in 1997. Since 1997, the French survey has been carried out on the R/V THALASSA, a stern trawler of 73.7 m length by 14.9 m width, and with gross tonnage of 3022 t. Only the time-series from 1997 was considered in this study. The fishing gear used is a GOV 36/47 with the exocet kite replaced by 6 additional floats. On average, the gear has a horizontal opening of 20 m and a vertical opening of 4 m. The doors are plane-oval and weigh 1350 kg. From 1997 onwards the whole area has been separated into 5 geographical strata or sectors: southern Bay of Biscay (GS) and northern Bay of Biscay (GN), southern Celtic Sea (CS), central Celtic Sea (CC) and northern Celtic Sea (CN). In each sector a depth-stratified sampling strategy has been adopted with 7 depth strata: 0–30, 31–80, 81–120, 121–160, 161–200, 201–400 and 401–600 meters.

Spanish Iberian northern shelf survey region

Surveys on the Northern Spanish Shelf started in 1984 and were undertaken in both spring and autumn up until 1988 when the spring survey ceased. The autumn survey is carried out on board R/V *Cornide de Saavedra*. In 1987 no survey was conducted. The gear used is a Baka trawl 44/60 with a 43.6 m footrope and a 60.1 m headline. The traditional trawl doors used are rectangular, weighting 650 kg with 3.6 m² of surface (2.67*1.34 m). The diameter of warp used is 22 mm (1.9 kg/m). The mean vertical opening is 1.8 m and the horizontal opening is 21 m. A codend cover of 20 mm mesh was used, and since 1985 a 20 mm mesh codend liner has been adopted. During the first eight years of the series (1984–1992) fish length was recorded for selected commercial species only. Therefore, only data since 1992 have been analysed.

Portuguese Iberian shelf and upper slope survey region

Since 1979 annual autumn groundfish surveys have been conducted on the Portuguese continental shelf in an area extending from latitude 41°20' N to 36°30' N (ICES Division IXa) and in waters from 20 m to 750 m in depth. No survey was undertaken in 1984. The survey uses a random stratified design (Cardador *et al.*, 1997) and since 1989 the 97 stations have been at fixed locations spread over 12 sectors, each divided into 4 depth ranges: 20–100 m, 101–200 m, 201–500 m and the 501–750 m; a total of 48 strata. Surveys have generally been carried out using the Portuguese 495GRT, 47.5 m, 1500 hp stern trawler RV "NORUEGA" deploying a Norwegian Campelen Trawl 1800/96 NCT bottom trawl with a 20 mm codend mesh size and bobbin ground-gear. In 1996, 1999, 2003 and 2004, the RV "CAPRICORNIO", deploying a FGAV019 bottom trawl without rollers in the ground-gear, was used instead (http://datras.ices.dk/). Because the LFI may be affected by such variation in survey methodology, these surveys in these years were excluded from our analyses.

Portuguese Azores archipelago demersal lines (DLL) survey

Data collected in spring from1996 to 2008 (except 1998 and 2006) by the longline groundfish survey conducted onboard RV "ARQUIPÉLAGO" off the Azores archipelago were analysed. The surveys, covering a region that included the nine islands and of some of the banks, followed a stratified random design based on geography and depth (50 m depth intervals) with about 30 fishing sets positioned in depths down to 600 m, at some stations down to a depth of 1200 m (Menezes *et al*., 2006). A standardized longline gear was used identical to the one used by commercial demersal fisheries in the Azores, locally known as a "stone/buoy long-line". It effectively targets bentho-pelagic species as well as benthic or demersal species with closer relations with the sea-bed (Melo, 1997). Since part of the gear floats off the bottom, due to the alternating buoys that are linked at the tying-rope edge with the main-rope, the gear design minimizes the risk of loss in the rough rocky bottom topography of this archipelago. Long-lines were set from four sided skates (each corresponding to a quarter-skate line), with about 30 hooks (hook size nº 9) per quarter-skate side, each approximately 36.5 m long. On average 12 skate gear lengths covered approximately 1 nautical mile. The bait used in all surveys was "chopped salted sardine", which is also often used by commercial fishing operations around the Azores (Menezes, 2003). Data collected from around Flores and Corvo islands were excluded from this analysis because these areas were not sampled in 1996 and 2008.

5.6.2.2 Results

Figure 5.6.2.2.1 summarizes trends in the LFI determined for four study areas. Trends were characterised by high inter-annual variability.

- Testing different methods to define large fish suggested that the indicator performed better when pelagic species were excluded and the effect of excluding pelagic schooling fish depended on the species excluded and the catchability of these species in the survey gear used. The southern Iberian shelf ecosystem, part of the Canary upwelling system, is dominated by small pelagic species. Exclusion of pelagic species in this region markedly affected the LFI, making it more sensitive to fishing impacts on the demersal fish community.
- Using different length thresholds did not substantially affect relative trends in the LFI. This suggests that the 40 cm threshold might also be suitable for defining the LFI in the SWW RAC area, but defining a suitable reference period and setting targets or limits for the LFI needs further work. Trends in the LFI were minimal, suggesting a relatively constant size composition within fish communities occupying the SWW RAC area. Setting an appropriate reference period for the LFI in sub-regions of the SWW RAC therefore requires reference to more historical information (Dinter, 2001), rather than using only these survey time-series, which may not go far enough back to a time when fishing in the area was sustainable. Alternatively, the lack of a trend may imply that current levels of fishing activity are sustainable.
- For some sub-regions in the SWW region attempts have been made to define unperturbed states using the earliest data available and by asking relevant experts to complete a survey in which three criteria were used to assess the state of fish communities: a) proportion of fully/overexploited

stocks >0; b) industrialized or destructive fishing practices were already prevalent in the area; and c) community or ecosystem impacts caused by fishing had been documented (Bundy *et al*., 2010).

To better understand LFI performance in each sub-region within the SWW RAC further information on the actual species and size composition of fish needs to be determined from the data available and better information on variation in levels of fishing pressure are needed.

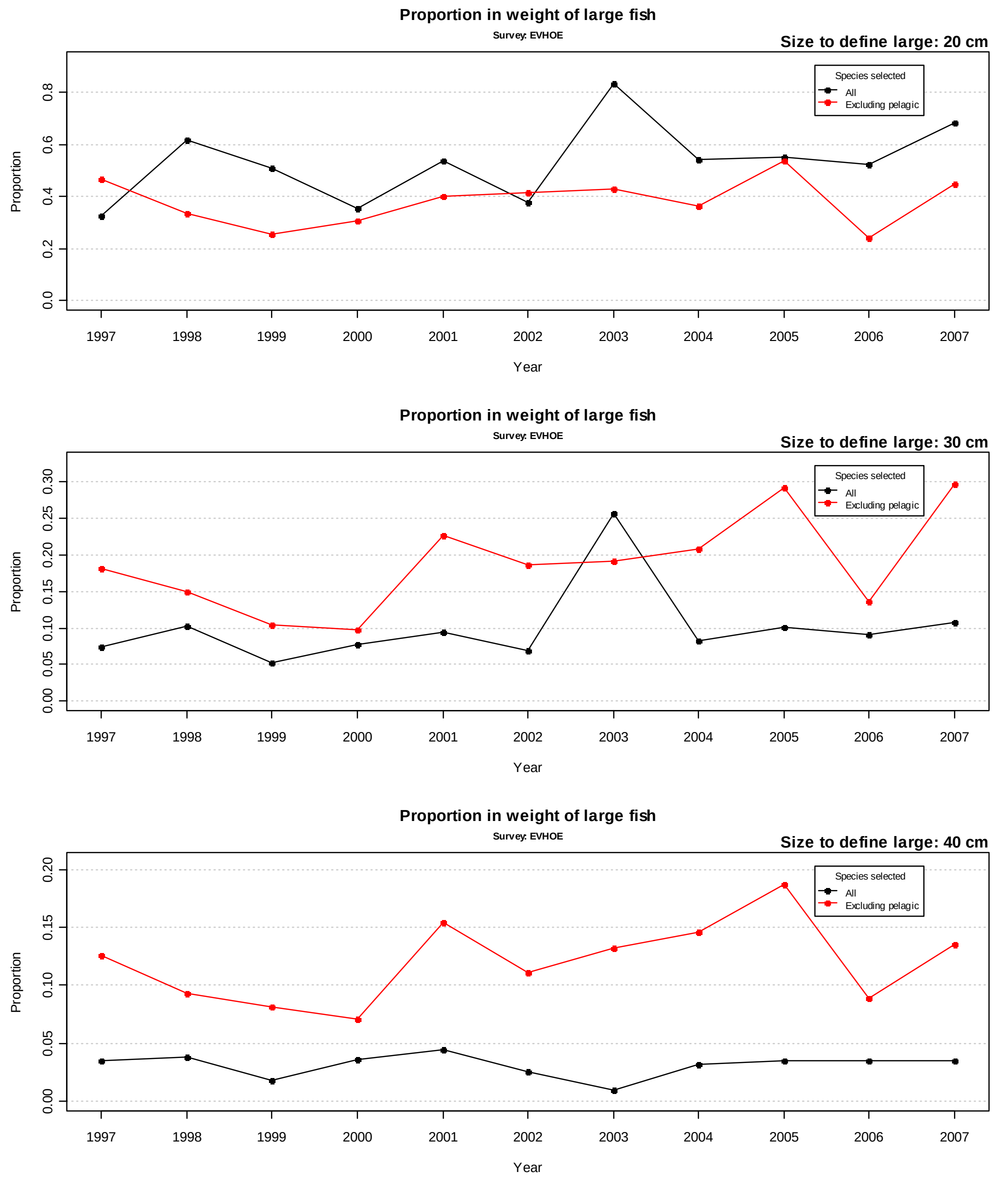


Figure 5.6.2.2.1. SWW area. LFI variation using three different length thresholds (top to bottom: 20 cm, 30 cm, 40 cm) to define large fish, and including (black line) and excluding (red line) pelagic species from the analysisof data from the Bay of Biscay area covered by the EVHOE survey (1997 to 2007).

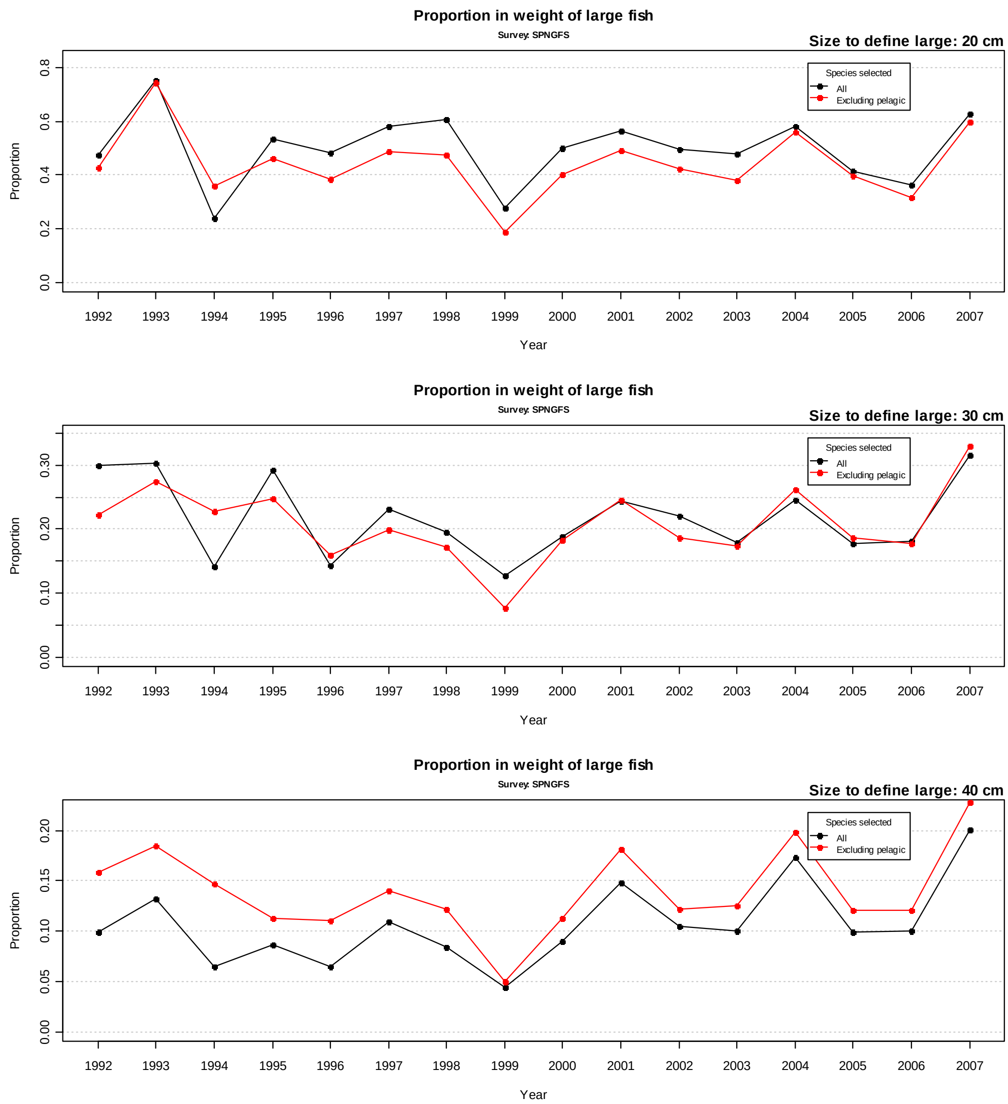
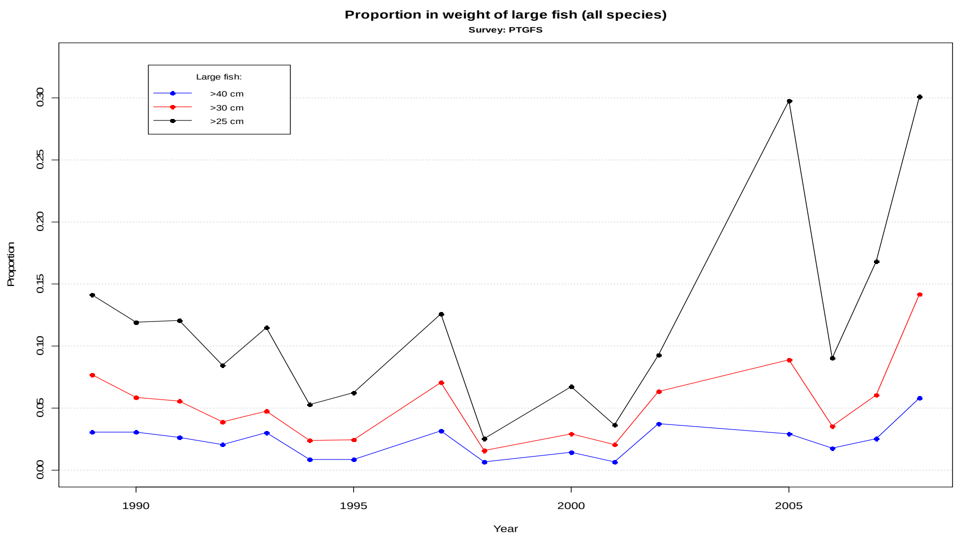


Figure 5.6.2.2.1. (cont.) SWW area. LFI variation using three different length thresholds (top to bottom: 20 cm, 30 cm, 40 cm) to define large fish, and including (black line) and excluding (red line) pelagic species from the analysis of the Northern Spanish shelf bottom trawl survey data (1992 to 2007).



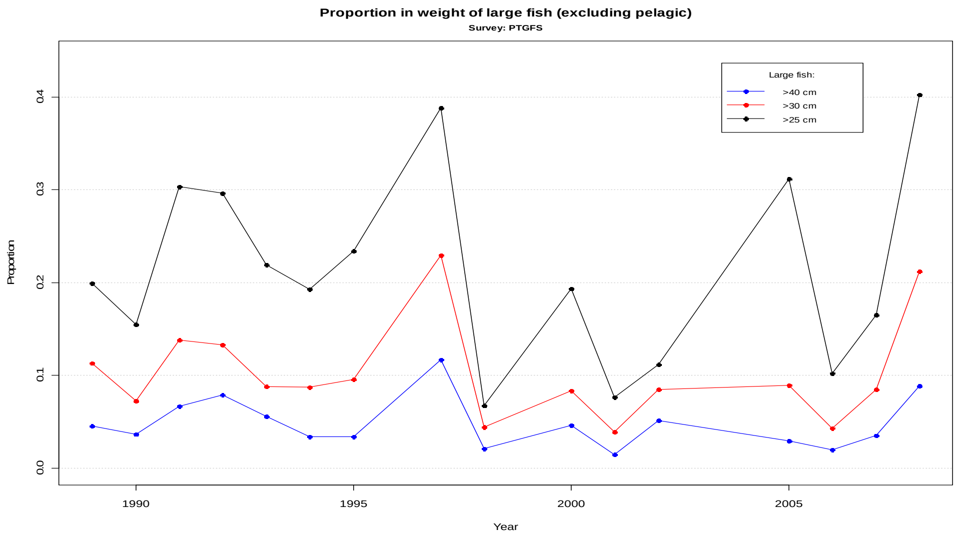


Figure 5.6.2.2.1. (cont.) SWW area. LFI variation using three different length thresholds (blue line refers to >40 cm size limit, the red line to >30 cm limit and the black line to >25 cm limit) to define large fish, and including (top figure) and excluding bottom figure) pelagic species from the analysis of the Portuguese bottom trawl survey data (1991 to 2008).

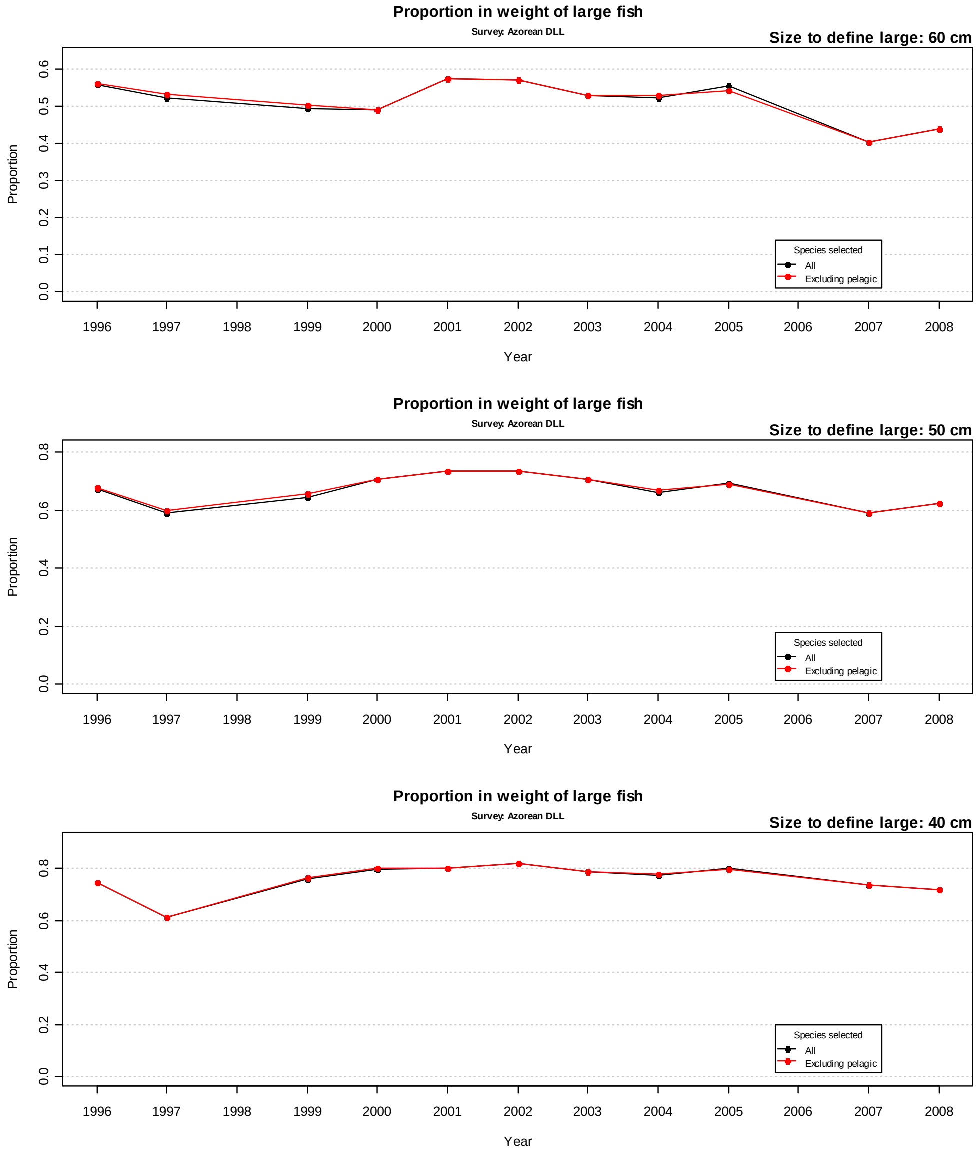


Figure 5.6.2.2.1. (cont.) SWW area. LFI variation using three different length thresholds (top to bottom: 60 cm, 50 cm, 40 cm) to define large fish, and including (black line) and excluding (red line) pelagic species from the analysis of the area covered by the Azores demersal long line survey (1996–2008).

5.6.3 Georges Bank, northeastern Atlantic

A preliminary analysis of data from Georges Bank, northeastern Atlantic was performed during the meeting. The large fish indicator (LFI) was calculated from the Northeast Fisheries Science Centre autumn trawl-survey data. The data were selected from survey strata on Georges Bank for the years 1963–2007. Pelagic species were excluded, leaving 50 demersal fish species. The stratified mean weight per tow was corrected for species-specific catchability and scaled to the area of Georges Bank to be in units of million tonnes. As for the North Sea, the LFI was calculated as the mass of fish greater than 40 cm length divided by the total mass in the corresponding year.

The large-fish biomass tracked the total biomass quite closely (Figure 5.6.3.1). Biomass varied without overall trend during the 45-year period. The troughs in biomass correspond to periods of intense foreign fishing around 1970 and intense domestic fishing in the early 1990s; conversely, the peaks correspond to periods of less-intense exploitation. Ranging between 0.45 and 0.8, the value of the LFI is high relative to other regions, as is expected from comparisons of the respective size spectra. The LFI is relatively less variable than biomass and appears to track the decade-scale biomass fluctuations.

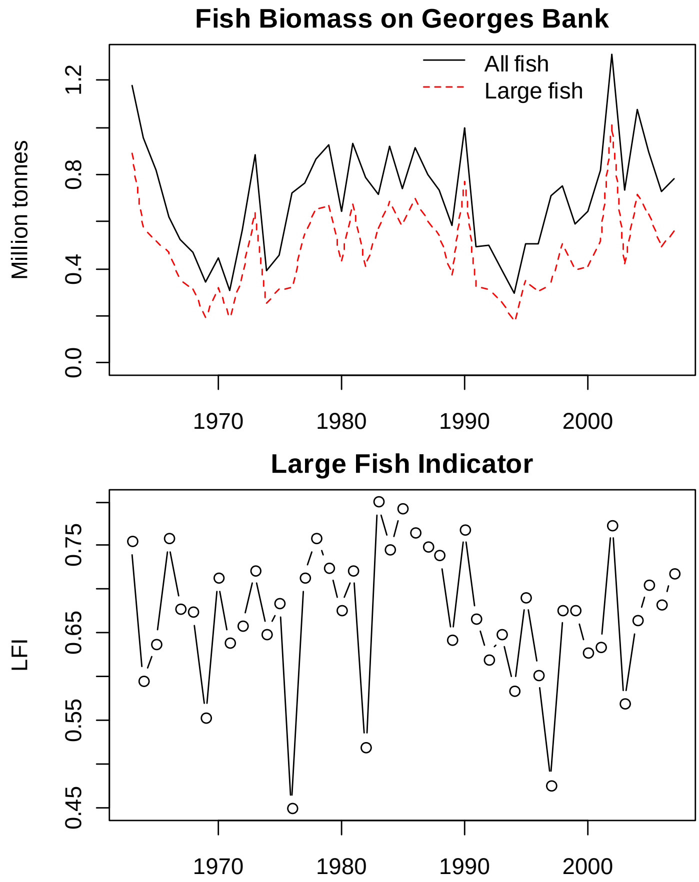


Figure 5.6.3.1. Biomass of large fish (>40 cm) on Georges Bank, calculated from Northeast Fisheries Science Center autumn trawl-survey data.

5.7 A need for protocols for determining the LFI

The LFI was developed over some years by WGFE and WGECO, and based principally on North Sea trawl survey data. Given that the LFI metric is now recommended in the DCF, and is being implemented outside the North Sea, WGECO recognises the need to provide guidance on how the "tuning" of the LFI was done in the North Sea, and on how it might be done in other sea areas.

Critical areas to address would be:

- Choice of species suite – e.g., pelagic species were not used in the North Sea, but this may not be appropriate elsewhere;
- Choice of cut-off level. Extensive study suggested the 40 cm level, but again this may well be incorrect in other ecosystems;
- Choice a weight based or number based metric. In the North Sea the metric is weight based, but number based approaches were also considered.

It is suggested that this would be a valuable contribution from WGECO to the wider community, and could be included as a ToR for WGECO in 2011.

5.8 References

Beverton, R. J. H. and Holt, S. J. 1957. On the dynamics of exploited fish populations. Fishery Investigations Series II, 19: 1–533.

Borges M.F., Velasco F., Mendes H. , Pinho M. R., Silva C., Porteiro C., Frid C.L.J., Paramor O.A.L., Piet G.J., Rogers S.I. Le Quesne W.J.F. 2010. Assessing the impact of fishing on the Marine Strategy Framework Directive objectives for Good Environmental Status. Developing and testing the process across selected RAC regions: The South Western Waters Region. Project Report Making European Fisheries Ecosystem Plans Operational (MEFEPO) EC FP7 project # 212881.

Bundy, A., Shannon, L. J., Rochet, M-J., Neira, S., Shin, Y-J., Hill, L., and Aydin, K. 2010. The good(ish), the bad, and the ugly: a tripartite classification of ecosystem trends. – ICES Journal of Marine Science, 67: 745–768.

Cardador, F., Sanchez, F., Pereiro, F. J., Borges, M. F., Caramelo, A. M., Azevedo, M., Silva, A., Perez, N., Martins, M. M., Olaso, I., Pestana, G., Trujillo, V., and Fernandez, A. 1997. Groundfish surveys in the Atlantic Iberian waters (ICES Divisions VIIIC and XIa): history and perspectives. ICES CM 1997/Y: 08, 30pp.

Carpenter, S. R., Kitchell, J. F., Hodgson, J. R., Cochran, P. A., Elser, J. J., Lodge, D. M., Kretchmer, D., He, X. and von Ende, C. N. 1987. Regulation of lake primary productivity by food web structure. Ecology, 68: 1863–1876.

Daan, N., Bromley, P.J., Hislop, J.R.G and Nielsen, N.A. 1990. Ecology of North Sea fish. Netherlands Journal of Sea Research, 26: 343–386.

Daan. N., Gislason, H., Pope, J. G. and Rice, J. C. 2005. Changes in the North Sea fish community: evidence of indirect effects of fishing. ICES Journal of Marine Science, 62: 177–188.

Dinter, WP. 2001. Biogeography of the OSPAR Maritime Area. A synopsis and synthesis of biogeographical distribution patterns described for the North-East Atlantic. Bundesamt für Naturschutz, Bonn, Germany, 167 pp.

Ehrich, S., Stelzenműller, V. and Adlerstein, S. 2009. Linking spatial pattern of bottom fish assemblages with water masses in the North Sea. Fisheries Oceanography, 18: 36–50.

Fraser, H. M., Greenstreet, S. P. R. and G. J. Piet. 2007. Taking account of catchability in groundfish survey trawls: implications for estimating demersal fish biomass. ICES Journal of Marine Science, 64:1800–1819.

Fraser, H. M., Greenstreet, S. P. R., Fryer, R. J., and Piet, G. J. 2008. Mapping spatial variation in demersal fish species diversity and composition in the North Sea: accounting for species- and size-related catchability in survey trawls. ICES Journal of Marine Science, 65: 531–538.

Greenstreet, S. P. R. 1996. Estimation of the daily food consumption of food by fish in the North Sea in each quarter of the year. Scottish Fisheries Research Report No.55.

Greenstreet, S.P.R., Fraser, H.M., Cotter, J., and Pinnegar, J. 2009. Assessment of the "state" of the demersal fish communities in OSPAR Regions II, III, IV, and V. Scottish Marine and Freshwater Science Report 00/09, pp. In Press.

Greenstreet, S. P. R., Holland, G. J., Fraser, T. W. K. and Allen, V. J. 2009. Modelling demersal fishing effort based on landings and days absence from port, to generate more useful indicators of "activity". ICES Journal of Marine Science, 66: 886–901.

Greenstreet, S. P. R. and Rogers, S. I. 2000. Effects of fishing on non-target fish species. Effects of Fishing on Non-Target Species and Habitats: Biological, Conservation and Socio-economic Issues. (eds M. J. Kaiser and B. de Groot), pp. 217–234. Blackwell Science, Oxford, U.K.

Greenstreet, S.P.R., Rogers, S.I., Rice, J.C., Piet, G.J. and Guirey, E.J. In Press. Development of the EcoQO for fish communities in the North Sea. – ICES Journal of Marine Science.

Guirey, E. J., Speirs, D., Gurney, W., Heath, M. and Greenstreet, S. P. R. 2008. A new multispecies, physiologically structured fish ecosystem model. ICES CM 2008/F:04.

Halpern, B. S., Walbridge, S., Selkoe, K. A., Kappel, C. V., Micheli, F., D'Agrosa, C., Bruno, J. F., Casey, K. S., Ebert, C., Fox, H. E., Fujita, R., Heinemann, D., Lenihan, H. S., Madin, E. M. P., Perry, M. T., Selig, E. R., Spalding, M., Steneck, R. and Watson, R. 2008. A global map of human impact on marine ecosystems. Science, 319: 948–952.

Heslenfeld, P. and Enserink, E. L. 2008. OSPAR Ecological Quality Objectives: the utility of health indicators for the North Sea. ICES Journal of Marine Science, 65: 1392–1397.

ICES, 2007a. Report of the Working Group on Assessment of demersal stocks in the North Sea and Skagerrak – Combined Spring and Autumn (WGNSSK), May 1–8 2007. ICES CM 2007\ACFM:18 and 30. 906 pp.

ICES, 2007b. Report of the Working Group on Fish Ecology (WGFE), 5–9 March 2007, Nantes, France. ICES CM 2007/LRC:03. 217 pp.

ICES, 2008a. Report of the Working Group on Fish Ecology (WGFE), 3–7 March 2008, Copenhagen, Denmark. ICES CM 2008/LRC:04. 119 pp.

ICES, 2008b. Report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO), May 6–13 2008, Copenhagen, Denmark. ICES CM 2008/ACOM:41. 269 pp.

ICES, 2009a. Report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO), 15–21 April 2009, Copenhagen, Denmark. ICES CM 2009/ACOM:20. 190 pp.

ICES, 2009b. Report of the Working Group on Fish Ecology (WGFE), 26–30 October 2009, Copenhagen, Denmark. ICES CM 2009/LRC:08. 133 pp.

Jennings, S., Greenstreet, S. P. R. and Reynolds, J. 1999. Structural change in an exploited fish community: a consequence of differential fishing effects on species with contrasting life histories. Journal of Animal Ecology, 68: 617–627.

Lee, J., South, A. B., and Jennings, S. 2010. Developing reliable, repeatable, and accessible methods to provide high-resolution estimates of fishing-effort distributions from vessel monitoring system (VMS) data. – ICES Journal of Marine Science, doi:10.1093/icesjms/fsq010.

Lekve, K., Ottersen, G., Stenseth, N. C. and Gjosaeter, J. 2002. Length dynamics in juvenile coastal Skagerrak cod: effects of biotic and abiotic factors. Ecology, 83: 1676–1688.

Mackinson, S. and Daskalov, G. 2007. An ecosystem model of the North Sea to support an ecosystem approach to fisheries management: description and parameterisation. Cefas Science Series Technical Report 142.

Melo, O. 1997. Distribuição vertical das capturas e Competição inter-especifica no palangre de fundo (pedra-bóia). Relatório de estágio, Porto, 68 pp.

Menezes, GM. 2003. Demersal fish assemblages in the Atlantic Archipelagos of the Azores, Madeira and Cape Verde. PhD thesis, Universidade dos Açores, Horta, 227p + appendix.

Menezes GM, Singles MF, Silva HM, Pinho, MR. 2006. Structure and zonation of demersal fish assemblages off the Azores Archipelago (mid-Atlantic). Mar. Ecol. Prog. Ser. 324:241–260.

Molloy, P. P., McLean, I. B. and Côté, I. M. 2009. Effects of marine reserve age on fish populations: a global meta-analysis. Journal of Applied Ecology, 46: 743–751.

Shurin, J. B. and Seabloom, E. W. 2005. The strength of trophic cascades across ecosystems: predictions from allometry and energetics. Journal of Animal Ecology, 74: 1029–1038.

Speirs, D. C., Guirey, E. J., Gurney, W. S. C. and Heath, M. R., submitted. A length-structured partial ecosystem model for cod in the North Sea. Fisheries Research.

Stelzenműller, V., Rogers, S. I., and Mills, C. M. 2008. Spatio-temporal patterns of fishing pressure on UK marine landscapes, and their implications for spatial planning and management. ICES Journal of Marine Science, 65: 1081–1091.

6 ToR d) Review methods used to determine "good environmental status"

d) Review methods used to determine "good environmental status" under the WFD, HD and MSFD, including a discussion of reference points and indicators;

6.1 Introduction

Europe now has several pieces of legislation that provide guidance and/or set standards for the quality of the environment that must be achieved or maintained in all or some parts of marine and coastal waters. In addition the Regional Seas Organizations have additional guidelines, standards and policies which their members are to apply. The pieces of legislation and other guidance differ in their high level goals, and correspondingly across the different Directives and Regional Seas Guidelines the standards to be met are not identical.

For this term of reference we undertake yet another cross-comparison of three major pieces of EU legislation that deal in setting standards for European marine ecosystems and/or managing activities that may affect environmental quality, as well as some other policy and guidance documents produced by Regional Seas bodies. The main pieces of legislation are the Water Framework Directive (WFD) (2000/60/EC), the Habitats Directive (HD) (92/43/EEC) and Marine Strategy Framework Directive (MSFD)(2008/56/EC). The central focus of the comparison was to place the new MSFD, which has the greatest scope of application, in context with the other Directives, guidelines and standards. Paragraph 8 of Article 1 of the MSFD states that:

"*By applying an ecosystem-based approach to the management of human activities while enabling a sustainable use of marine goods and services, priority should be given to achieving or maintaining good environmental status in the Community's marine environment, to continuing its protection and preservation, and to preventing subsequent deterioration.*"

Moreover in Paragraph 25 of Article 1 it states that Member States should then determine for their marine waters a set of characteristics for good environmental status. For those purposes, it is appropriate to make provision for the development of criteria and methodological standards to ensure consistency and to allow for comparison between marine regions or subregions of the extent to which good environmental status is being achieved.

How the concepts of "a sustainable use", "good environmental status" (GES), and "ensure consistency and to allow for comparison between marine regions and subregions" are interpreted will be crucial to implementation of of the MSFD. The MSFD leaves scope for States to interpret and apply "sustainable use" and GES, but the concepts both have a science basis, and the requirement for consistency adds an additional science role in informing the decisions about whether or not marine areas that intrinsically different are still "consistent" in their environmental status (See Section 3.3). Annexes 1 and 3 of the Directive contain some guidance on interpretation, in the specification of a set of "Descriptors" of GES, and pressures that need to be considered.

ICES, in conjunction with the Joint Research Council, has already provided substantial science input to support the interpretation and application of the provisions of the MSFD regarding GES (Cardosa *et al.*, 2010). This support has been taken up by the EC in their "Elements of Decision" (EC, 2010). However, in discussions at the EU working Group on Good Environmental Status, within States, and in ICES, it is clear that future science advice is needed for successfully interpreting and applying these concepts. The national and WGGES dialogue has also made clear that the positioning of the requirements of the MSFD relative to the requirements of existing Directives is of concern for several reasons. One set of reasons are efficiencies: States wish to ensure that investments made in monitoring, assessments, and regulation to support of implementing other Directives and Guidelines are used to the fullest extent possible implementing the MSFD. Another set of reasons are concerns about harmonizing the standards that are to be achieved among the Directives and Guidelines, to regulatory authorities: to what extent do measures implemented for one Directive aid or work in conflict with measures needed under the new MSFD?

In Section 6.2 we provide additional guidance on these issues. For each major Directive we extracted:

- The geographic coverage of the Directive/Guidelines and ways to opt areas in or out;
- The higher level objectives of the Directive/Guidelines and the reference levels that from the objectives;
- The assessment process used (or proposed), and the process used to select the indicators and methods to combine information from the individual indicators into the evaluation of status;
- The approaches used to achieve consistency in the evaluations of status under the Directives/guidelines across regions and subregions.

From this information Section 6.3 lays out proposals for a way forward with implementation of the MSFD that is scientifically sound, while being as efficient as practical in demands for monitoring and assessment, and harmonious with implementation of other Directives.

6.2 Comparison of Water Framework Directive, Habitat Directive, and Marine strategy Framework Directive

6.2.1 Water Framework Directive (WFD)

The Water Framework Directive (2000/60/EC hereinafter referred to as "WFD"; later amended by Decision No 2455/2001/EC, Directive 2008/32/EC) establishes a framework for Community action in the field of water policy aiming at:

i) preventing further deterioration, protect and enhance the status of aquatic ecosystems;

ii) promoting sustainable water use;

iii) enhancing protection and improvement of aquatic environment;

iv) ensuring the progressive reduction of pollution of groundwater;

v) mitigating the effects of floods and droughts.

6.2.1.1 Spatial domain

The WFD applies to surface and ground waters. Surface waters are distinguished into: i) rivers; ii) lakes; iii) transitional waters (i.e., bodies of surface water in the vicinity of river mouths which are partly saline in character as a result of their proximity to coastal waters but which are substantially influenced by freshwater flows); and iv) coastal waters (i.e., surface water on the landward side of a line, every point of which is at a distance of one nautical mile on the seaward side from the nearest point of the baseline from which the breadth of territorial waters is measured, extending where appropriate up to the outer limit of transitional waters) (Art. 2, WFD).

Moreover, the WFD contains prescriptions related to "artificial water body" (i.e., a body of surface water created by human activity) and "heavily modified water body (i.e., a body of surface water which as a result of physical alterations by human activity is substantially changed in character) which are designated by the Member State in accordance with the provisions of Annex II of the Directive.

Individual river basins lying within the national territory of each Member State, along with small river basins, ground waters and coastal waters must be assigned to individual river basin districts (Art. 3.1, WFD). River basins covering the territory of more than one Member State are assigned to an international river basin district and Member States must ensure the appropriate administrative arrangements for the application of the rules of this Directive within the portion of any international river basin district lying within its territory (Art. 3.3, WFD). Where a river basin district extends beyond the territory of the Community, the Member State or Member States concerned shall endeavour to establish appropriate coordination with the relevant non-Member States, with the aim of achieving the objectives of the WFD throughout the river basin district. Such arrangement was for instance accomplished, involving Bulgaria, Norway and Romania in the intercalibration exercise (2005/646/EC).

6.2.1.2 What are the ecological standards/reference levels that are set?

Considering surface waters, the general objective of the WFD is to achieve the 'good status' by 2015 in all Member States. 'Good status' is defined as the status achieved by a surface water body when both its ecological status (i.e., an expression of the quality of the structure and functioning of aquatic ecosystems, classified in accordance with Annex V of the WFD) and its chemical status (concentrations of pollutants compared with the environmental quality standards established in Annex IX and under Article 16(7) of the WFD, and under other relevant Community legislation setting environmental quality standards at Community level) are at least 'good'.

Different prescriptions applies to artificial and heavily modified water bodies where the target is to achieve the 'good ecological potential' (as defined in Anne V of the Directive) and good surface water chemical status at the latest 15 years from the entry into force of the WFD.

6.2.1.3 Assessment of water status

The WFD classification scheme for water quality includes five status classes: high, good, moderate, poor and bad.

'High status' is defined as the biological, chemical and morphological conditions associated with no or very low human pressure. This is also called the 'reference condition' as it is the best status achievable - the benchmark. These reference conditions are type-specific, so they are different for different types of rivers, lakes or coastal waters so as to take into account the broad diversity of ecological regions in Europe.

Assessment of quality is based on the extent of deviation from these reference conditions, following the definitions in the Directive. 'Good status' means 'slight' deviation, 'moderate status' means 'moderate' deviation, and so on.

The definition of ecological status takes into account specific aspects of the biological quality elements (i.e., phytoplankton, aquatic flora, benthic invertebrate fauna, fish fauna), for example "composition and abundance of aquatic flora" or "composition, abundance and age structure of fish fauna" (see Table 6.2.1.3.1; WFD Annex V Section 1.1 for the complete list). These definitions are expanded in Annex V to the WFD.

Table 6.2.1.3.1. Quality elements to be used for the assessment of ecological status/potential based on the list in Annex V, 1.1, of the WFD.

Annex V 1.1.1. RIVERS	**Annex V 1.1.2. LAKES**	**Annex V 1.1.3. TRANSITIONAL WATERS**	**Annex V 1.1.4. COASTAL WATERS**
BIOLOGICAL ELEMENTS			
• *Composition and abundance of aquatic flora*[3] • *Composition and abundance of benthic invertebrate fauna* • *Composition, abundance and age structure of fish fauna*	• *Composition, abundance and biomass of phytoplankton* • *Composition and abundance of other aquatic flora*[4] • *Composition and abundance of benthic invertebrate fauna* • *Composition, abundance and age structure of fish fauna*	• *Composition, abundance and biomass of phytoplankton* • *Composition and abundance of other aquatic flora*[5] • *Composition and abundance of benthic invertebrate fauna* • *Composition and abundance of fish fauna*	• *Composition, abundance and biomass of phytoplankton* • *Composition and abundance of other aquatic flora*[5] • *Composition and abundance of benthic invertebrate fauna*
HYDROMORPHOLOGICAL ELEMENTS SUPPORTING THE BIOLOGICAL ELEMENTS			
• *Hydrological regime* → *quantity and dynamics of water flow* → *connection to ground water bodies* • *River continuity* • *Morphological conditions* → *river depth and width variation* → *structure and substrate of the river bed* → *structure of the riparian zone*	• *Hydrological regime* → *quantity and dynamics of water flow* → *residence time* → *connection to the ground water body* • *Morphological conditions* → *lake depth variation* → *quantity, structure and substrate of the lake bed* → *structure of the lake shore*	• *Tidal regime* → *freshwater flow* → *wave exposure* • *Morphological conditions* → *depth variation* → *quantity, structure and substrate of the bed* → *structure of the intertidal zone*	• *Tidal regime* → *direction and dominant currents* → *wave exposure* • *Morphological conditions* → *depth variation* → *structure and substrate of the coastal bed* → *structure of the intertidal zone*
CHEMICAL AND PHYSICOCHEMICAL ELEMENTS SUPPORTING THE BIOLOGICAL ELEMENTS			
• *General* → *thermal conditions* → *oxygenation conditions* → *salinity* → *acidification status* → *nutrient conditions* • *Specific pollutants* → *pollution by priority substances identified as being discharged into the body of water* → *pollution by other substances identified as being discharged in significant quantities into the body of water*	• *General* → *transparency* → *thermal conditions* → *oxygenation conditions* → *salinity* → *acidification status* → *nutrient conditions* • *Specific pollutants* → *pollution by priority substances identified as being discharged into the body of water* → *pollution by other substances identified as being discharged in significant quantities into the body of water*	• *General* → *transparency* → *thermal conditions* → *oxygenation conditions* → *salinity* → *nutrient conditions* • *Specific pollutants* → *pollution by priority substances identified as being discharged into the body of water* → *pollution by other substances identified as being discharged in significant quantities into the body of water*	• *General* → *transparency* → *thermal conditions* → *oxygenation conditions* → *salinity* → *nutrient conditions* • *Specific pollutants* → *pollution by priority substances identified as being discharged into the body of water* → *pollution by other substances identified as being discharged in significant quantities into the body of water*

6.2.1.4 Methods to ensure consistency

The WFD includes a harmonisation and calibration exercise. This involves the use of reference sites chosen by national authorities to be at the boundaries of high/good status, and good/moderate status. The intercalibration exercise (Annex V, 1.4.1, iii) then sets out to achieve consistency between these boundaries (thresholds) within geographical areas (see below). There is no explicit plan to further harmonise between these geographical areas. Guidelines regarding the intercalibration process are provided in the Technical report on the Water Framework Directive intercalibration exercise.

In this framework an intercalibration network, consisting of selected sites, has been established representing Member States' interpretations of the normative definitions of surface water status (defined in Annex V, Section 1.2) in relation to reference conditions (2005/646/EC).

Selected sites are classified according to the following information:

- water category;
- Geographical Intercalibration Group (GIG);
- boundary the site most closely represents (high-good (HG) or good-moderate (GM), according to Member State's assessment of the ecological quality status.

The main goal of the intercalibration exercise is to ensure comparable ecological quality assessment systems and harmonised ecological quality criteria for surface waters in the Member States. This ensures a harmonised approach to define one of the main environmental objectives of the WFD, the "good ecological status", by establishing:

- Agreed ecological quality criteria for good quality sites, setting the targets for protection and restoration;
- Agreed numerical Ecological Quality Ratio (EQR) values for two quality class boundaries (high/good and good/moderate).

Intercalibration is carried out by the Member States in the framework of Geographical Intercalibration Groups (GIG), that have been set up in order to ensure intercalibration across similar surface body types according to the Ecoregions to which they belong, thus providing either a geographical and ecological consistency to this framework..

The GIG referred to transitional and coastal waters defined in the Commission Decision (2005/646/EC) are given in Table 6.2.1.4.1.

Table 6.2.1.4.1. Transitional and coastal waters: intercalibration groups and participating Member States.

Intercalibration Group	Participating Member States
Baltic (CBA)	Denmark, Estonia, Finland, Germany, Latvia, Lithuania, Poland, Sweden
North-East Atlantic (CNE)	Belgium, Denmark, France, Germany, Ireland, Netherlands, Portugal, Spain, Sweden, United Kingdom
Mediterranean (CME)	Cyprus, France, Greece, Italy, Malta, Slovenia, Spain

Although the WFD defines which biological elements must be taken into account when assessing ecological status, it leaves the Member States flexible to define the details of their own assessment system. That is why the purpose of intercalibration is not to harmonise assessment systems (e.g., to use the same indicator to derive quality criteria for good quality sites), but only their results.

This implies that, after the intercalibration, Member States sharing the same surface water body types within a GIG, will use different ecological quality ratios for defining the boundaries between ecological status classes (tailored to their classification system), but these values will discriminate between equivalent ecological status.

The results of the intercalibration process, which cover only part of the whole assessment, have been published in the Commission Decision 2008/915/CE and are expected to be integrated with another Decision by 2012.

Member States are bound to apply the results of the intercalibration process in the framework of the evaluation of good ecological status and targets of the WFD.

6.2.2 Habitats (and Species) and Birds Directives

These two Directives provide the means by which the EC enacts its commitments for nature conservation made under a number of international agreements.

6.2.2.1 Council Directive 79/409/EEC on the conservation of wild birds

The European Community meets its obligations for bird species under the Bern Convention and Bonn Convention by means of the Council Directive 79/409/EEC on the conservation of wild birds (commonly known as the 'Birds Directive'). The Directive provides a framework for the conservation and management of, and human interactions with, wild birds in Europe. It sets broad objectives for a wide range of activities, although the precise legal mechanisms for their achievement are at the discretion of each Member State.

The main provisions of the Directive include:

- The maintenance of the favourable conservation status of all wild bird species across their distributional range (Article 2) with the encouragement of various activities to that end (Article 3);
- The identification and classification of Special Protection Areas for rare or vulnerable species listed in Annex I of the Directive, as well as for all regularly occurring migratory species, paying particular attention to the protection of wetlands of international importance (Article 4);
- The establishment of a general scheme of protection for all wild birds (Article 5);
- Restrictions on the sale and keeping of wild birds (Article 6);
- Specification of the conditions under which hunting and falconry can be undertaken (Article 7). (Huntable species are listed on Annex II.1 and Annex II.2 of the Directive);
- Prohibition of large-scale non-selective means of bird killing (Article 8);
- Procedures under which Member States may derogate from the provisions of Articles 5–8 (Article 9) that is, the conditions under which permission may be given for otherwise prohibited activities;
- Encouragement of certain forms of relevant research (Article 10);
- Requirements to ensure that introduction of non-native birds do not threatened other biodiversity (Article 11).

6.2.2.2 Council Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora

In 1992 the European Community adopted Council Directive 92/43/EEC on the Conservation of natural habitats and of wild fauna and flora (commonly known as the EC Habitats Directive even though it provides protection for a range of species (listed in Annex II)). This is the means by which the Community meets its obligations as a signatory of the Convention on the Conservation of European Wildlife and Natural Habitats (Bern Convention). The provisions of the Directive require Member States to introduce a range of measures including the protection of species listed in the Annexes, and to undertake surveillance of habitats and species and produce a report every six years on the implementation of the Directive. The 189 habitats listed in Annex I of the Directive and the 788 species listed in Annex II, are to be protected by means of a network of sites. Each Member State is required to prepare and propose a national list of sites for evaluation in order to form a European network of Sites of Community Importance (SCIs). Once adopted, these are designated by Member States as Special Areas of Conservation (SACs), and along with Special Protection Areas (SPAs) classified under the EC Birds Directive, form a network of protected areas known as Natura 2000. The Directive was amended in 1997 by a technical adaptation Directive. The annexes were further amended by the Environment Chapter of the Treaty of Accession 2003.

The Habitats Directive introduces:

- Measures for the designation of protected areas;
- The precautionary principle; that is that projects can only be permitted having ascertained no adverse effect on the integrity of the site. Projects may still be permitted if there are no alternatives, and there are imperative reasons of overriding public interest. In such cases compensation measures will be necessary to ensure the overall integrity of network of sites.
- As a consequence of amendments to the Birds Directive these measures are to be applied to SPAs also.
- Member States shall also endeavor to encourage the management of features of the landscape to support the Natura 2000 network.

6.2.2.3 To what areas (spatial) does it apply?

The Habitats (92/43/EEC) and Birds (79/409/EEC) Directives apply to the whole European territory. EU Member States have an obligation to apply nature legislation in waters under their jurisdiction and, outwards, in waters where they exercise sovereign rights. The Habitats and Birds Directives apply in the European territory of the Member States[2] . The overall aim on land and on sea of both directives is to maintain or restore natural habitats and species of wild fauna and flora of community interest at a favourable conservation status. Establishing the European network of protected areas "Natura 2000" is one of the most important instruments to fulfil the task of the two European Nature Directives. Sites protected under the Habitats Directive are classed as sites of Community importance (SCIs) and as special areas of conservation (SACs). Areas protected under the Birds Directive are described as special protection areas (SPAs).

6.2.2.3.1 Sites protected under the Birds Directive

Once Member States propose sites to the EU Commission for protection under the Birds Directive, the sites automatically become special protection areas (SPAs) and are included in the EU's Natura 2000 network of protection areas. SPAs are selected for the especially endangered bird species listed in Annex I of the Birds Directive. Under Article 4 (1) of the Directive, the 'most suitable territories in number and size' should be declared as special protection areas. Article 4 (2) requires that Member States shall take similar measures for regularly occurring migratory species not listed in Annex I, bearing in mind their need for protection in the geographical sea and land

[2] Guidelines for the establishment of the Natura 2000 network in the marine environment. Application of the Habitats and Birds Directives .

area where this Directive applies, as regards their breeding, moulting and wintering areas and staging posts along their migration routes.

For each site, specific information must be provided along with cartographic material in analogue and digital form. These requirements are governed by Commission Decision (97/266/EC) of 18 December 1996 concerning a form to provide information about sites proposed for inclusion in the Natura 2000 network. The Standard Data Form developed for the purpose requires details of the site code (name, size, etc.), the site location and a brief description of its importance, vulnerability, protection status, and management and conservation objectives. Apart from mandatory information, optional details may also be given. A key component of the information provided about a site involves details of the occurring bird species.

6.2.2.3.2 Sites protected under the Habitats Directive

Designation of sites protected under the Habitats Directive takes place in two stages. Selection of sites for inclusion in the Natura 2000 network is subject to the nature conservation-related criteria set out in Article 4 and Annex III of the Habitats Directive and determined by the species and habitat types listed in the Directive's Annexes I and II. Following a ruling by the European Court of Justice, neither political necessity nor economic nor infrastructural interests may play a role when selecting sites and setting their boundaries. Sites suitable for inclusion in the Natura 2000 network include those which contribute significantly to the maintenance or restoration at a favourable conservation status of natural habitat types or species listed in the Habitats Directive. Natura 2000 sites are also intended to contribute significantly to the coherence of the protected area network and to biodiversity in the biogeography regions within the European Union. For animal species ranging over wide areas, Sites of Community Importance correspond to the places within the natural range of such species that present the necessary physical and biological factors essential to their survival and reproduction.

The procedure for site selection involves two separate stages:

In stage one, the EU Member States identify proposed Sites of Community Importance (pSCIs) for inclusion in the Natura 2000 network and forward them to the European Commission together with supplementary data and maps.

In stage two, the sites are assessed at the EU level for their importance to the Community. A List of Sites of Community Importance is then drawn up by the EU Commission in conjunction with the Member States (Article 4 (2) of the Habitats Directive). Assessment of the pSCIs in both stages is conducted in accordance with the criteria set out in Annex III of the Habitats Directive – an assessment at national level in stage one and an assessment at EU level in stage two. The two assessments serve selection of the most suitable sites for inclusion in the List of Sites of Community Importance. Once the lists are complete, the Member States are required to comply with the provisions of national legislation to place their respective sites under protection as special areas of conservation (SACs) within a period of six years.

6.2.2.4 What are the favourable conservation standards/reference levels that are set?

The overall aim on land and on sea of the Habitats and Birds Directives is to maintain or restore natural habitats and species of wild fauna and flora of community interest at a favourable conservation status.

The 'conservation status' is seen as the result of influences which include the present state of the habitat, together with current environmental and human influences (both positive and negative), that may influence its long-term survival. The status will be taken as favourable when its natural range and areas it covers within that range are stable or increasing, the species structure and functions which are necessary for its long term maintenance exist and are likely to continue to exist for the foreseeable future, and the conservation status of its typical species is favourable.

The conservation status of species will be taken as favourable when population dynamics data on the species concerned indicate that it is maintaining itself on a long-term basis as a viable component of its natural habitats and there is and will continue to be a sufficiently large habitat to maintain its populations on a long-term basis.

The European Commission and Member States have agreed standards for classifying the status of these habitats and species. This is to ensure that all Member States report on a similar basis. When assessing the conservation status of habitats, four elements are considered. These are: range, area, structure and function (= habitat condition) and future prospects. For species, the elements are: range, population, habitat (extent and condition) and future prospects.

Each of these elements is assessed as being in one of the following conditions: favourable, unfavourable-inadequate, unfavourable-bad, or unknown. The European Commission and Member States have agreed standards for these assessments, and the European Commission has also produced supplementary guidance.

6.2.2.5 Assessment of status

The Habitats Directive legally distinguishes two cases in which the conservation status of habitats and species is expected to be assessed. Member States have to monitor and report regularly (every six years) to the commission according to Art. 11 and 17 of the Habitats Directive on the conservation status of the habitats and species regardless of whether they are situated inside a Natura 2000 site or outside. If the overall conservation status of a species or habitat in a given biogeographical region is not favourable, the conservation measures must be improved.

Member States assess the conservation status of species and habitat types separately for each biogeographic region. It is necessary to use the biogeographic regions as reference regions to take proper account of the overall European range with its biogeographical variations.

Additionally, all plans or projects that - individually or in combination with other plans or projects - are likely to have a significantly adverse effect on the integrity of a designated 'Natura 2000' site in the light of conservation objectives of habitats and species (Art. 6 (2) and (3)), should also be assessed.

Under the Birds Directive, Member States are required to prepare a composite report on the implementation of the Directive every 3 years (Article 12).

6.2.2.6 Methods to ensure consistency

At present there aren't any clear methods that ensure consistency between Member States. The Directive's criteria for the Favourable Conservation Status are rather general and can therefore not directly be applied for each and every particular species or habitat. Furthermore, the ecological requirement of one and the same species may vary depending on the physical, climatic and geographical circumstances in each member state. This implies that each country has to define its own criteria and set of parameters for assessing FCS based on national conditions and processes, which are linked to 1) natural distribution range, 2) typical structures and functions of the habitat types and the species' habitats, and 3) future prospects. Some countries such as the

UK (JNCC 2007) and Bulgaria (Zingstra *et al*., 2009) have already formulated some guidelines.

6.2.3 Marine Strategy Framework Directive

6.2.3.1 To what areas (spatial) does it apply?

In article 4 of the MSFD the marine regions and sub-regions are defined as 1) the Baltic Sea; 2) the Northeast Atlantic Ocean with the sub regions of the Greater North Sea, including the Kattegat, and the English Channel; the Celtic Seas; the Bay of Biscay and the Iberian Coast; in the Atlantic Ocean, the Macaronesian bio-geographic region, being the waters surrounding the Azores, Madeira and the Canary Islands; 3) the Mediterranean Sea with the sub-regions of the Western Mediterranean Sea; the Adriatic Sea; the Ionian Sea and the Central Mediterranean Sea; the Aegean-Levantine Sea; and 4) the Black Sea.

Although there are no spatial exceptions, the directive identifies reasons why Member States may not achieve the environmental targets or good environmental status, either completely or in the required timescale. Reasons include lack of responsibility for actions, natural causes, force majeure, modifications or alterations to the physical characteristics through overriding public interest, and natural conditions which do not allow timely improvement in the status of the marine waters concerned.

6.2.3.2 What are the environmental standards/reference levels that are set?

The overall goal of the MSFD is the application of an ecosystem-based approach to the management of human activities. This ensures 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. Thus priority should be given to achieving or maintaining good environmental status in the Community's marine environment, to continuing its protection and preservation, and to preventing subsequent deterioration.

While ecological and pressure-related descriptors of good environmental status are provided, the MSFD does not provide indicators or reference levels. It states that when devising them, Member States shall take into account the application of relevant existing national, Community or international level environmental targets, ensuring they are mutually compatible and address any transboundary impacts. Task Groups established by ICES and the JRC have developed more detailed proposals for each descriptor, suggesting potential indicators and ecological characteristics (from bad to good) across a pressure gradient. In the management group report (Cardoso *et al*., 2010) summarizing the output from these Groups, the importance of flexible reference levels is highlighted, especially as a response to changing baselines caused by environmental changes. However, there are obvious research gaps on how to identify adequate reference levels, and these will need to be addressed by Member States before full implementation of the Directive can be achieved.

The MSFD recognises that due to the transboundary nature of the marine environment, Member States should cooperate to ensure the coordinated development of marine strategies for each marine region or subregion and a subsequent standardized assessment. For this reason the ecological standards which relate to the sustainable use of the sea need to be assessed in a consistent and scientifically sound way. To agree on environmental standards within a shared Regional Sea, Member States should use existing institutional structures established in marine regions or subregions, in particular Regional Sea Conventions. However, all Regional Seas Conventions play a role in that process.

6.2.3.3 Assessment of status

Currently, the process by which Member States will choose the indicators for each descriptor is not clearly defined. Having selected the most appropriate indicators for the regional sea, it will then be necessary for Member States to combine the outcome to reach a decision on the status for each descriptor. There are three levels of integration required to move from evaluation of the individual indicators to an assessment of GES, these include the assessment of, a) Indicators within individual Attributes of a Descriptor (for complex Descriptors), b) Status across all the Attributes within a Descriptor, and c) Status across all Descriptors. Discussion of practical methodology at each of these stages has not yet taken place, and sets a major challenge for the Commission and Member States in the effective implementation of the Directive (see Section 6.3 for further elaboration on how to move forward with the MSFD).

6.2.3.4 Methods to ensure consistency

The spatial assessment unit of the Directive is the Region or Sub-Region. Member States are required to complete marine strategies that describe the characteristics of the environment (following an initial assessment), describe the major pressures and impacts, and social and economic features. The extent to which Member States cooperate will determine the consistent standards of assessment within these areas will be assured when Member States monitoring programmes. The Directive also makes it clear that, where practical and appropriate, existing institutional structures established in marine regions or subregions, such as the Regional Sea Conventions, should be used to ensure such coordination.

Member States select the final set of indicators and related reference levels to assess GES. To ensure consistency of the assessment across regions it is crucial to provide scientifically sound guidance on the process on how to define threshold values (see Section 6.3).

Member States are also required, 'in the interests of coherence and coordination' (Art 10) to ensure that monitoring methods are consistent across the marine region to allow comparison of results, and that relevant transboundary impacts and features are taken into account.

6.3 Moving forward with the MSFD

It is clear from Section 6.2 that there are both important similarities and important differences among the WFD, the HD, and the MSFD. It is the latter which will be the focal concern for this section of the Report. Section 6.2.3 makes clear that the process for evaluating GES is still under development. However, whatever process emerges for evaluating GES, it will have to meet several standards that the discussions in Sections 3 and 4 make clear are not easy to achieve. The assessments have to be sound, make best use of available information, and integrate information both across ecosystem components and between pressures and states. That in itself is not easy (Section 3.4 of this report) However, greater challenges are posed by the need for the evaluation process to have two additional characteristics. On one hand it will have to have substantial flexibility to accommodate both the variation among the ecosystems and their uses, and the substantial scope for choice offered to States for how they conduct their national assessments of GES. On the other hand the process will have to demonstrate consistency of ecological standards being applied across the evaluations, in the face of differences in the ecosystem components, pressures, indicators, and reference levels being used in the assessment.

The rest of this Section provides an analysis of the issues involved in developing an approach to assessing GES that is simultaneously scientifically sound, flexible, ensures consistency, and provides guidance for moving ahead on some of the larger science challenges to the developing process. Several members of WGECO were also members of Task Groups of the ICES-JRC initiative, so the ideas developed here are well informed by the ideas and conclusions of those Task Groups and their Management Group.

6.3.1 How to choose suites of indicators from the large candidate set

The guidance for this step is contained fully in Section 3.5 of this Report. Annexes I and III to the MSFD delineate the scope of indicators of ecosystem components and pressures that need to be considered in the evaluations of GES. The Elements of the Decision take a further step in listing a large set of candidate indicators and classes of indicators that are promoted for use in the GES evaluations. Section 3.5 of this Report has the guidance needed on how, on a regional, subregional, or ecosystem scale, sound choices can be made about which classes of indicators and specific indicators within the classes should be chosen or use in evaluating GES. In particular Sections 3.5 and 3.7 could contribute valuable information to any process that has to choose a balanced set of indicators that address adequately ecosystem structure, function, the levels of pressures, and the interactions of ecosystem components with diverse pressures. WGECO encourages that Section 3 of this report be considered by all groups working on selection of indicators.

6.3.2 How to set reference levels on the chosen indicators

Section 4 discusses the complexities of setting reference levels that are ecologically consistent both across diverse types of ecosystem properties and pressures, and across ecosystems that are themselves different. The first step in setting a reference level for an indicator is to decide what ecological property the reference level has to represent. This is rooted in the policy objective that the indicator is supposed to serve. It is often the case that the policy objective is not explicit enough about the ecological state intended to be achieved by the objective to make the choice of a reference quantitatively straightforward. For example "keep the community productive and diverse" or "achieve good environmental status" will require substantial interpretation to select a corresponding reference level on an indicator used to measure progress towards the objective. Even where a policy objective gives clear guidance on the nature of the corresponding reference level, an indicator may be applied in multiple places, say around the marine areas of Europe, or in all the waters of a Regional Seas organization. Among those places ecological conditions differ enough that a single absolute or relative value may not capture the desired ecological conditions in every place. Hence the science advisory process will have to apply sound and systematic reasoning to translate the ecological status intended by a policy directive into the ecological properties that should be captured by the corresponding reference level on an indicator. The science "translation process" should strive to ensure the reasoning applied is sound and transparent, so the reference levels that are selected are ecologically appropriate for both the policy and the ecological conditions in the area, and not *ad hoc* and arbitrary.

There are several extra complications when reference levels are set for specific areas:

- Sometimes a policy may include provisions regarding more than one desired (or at least benchmark) ecological condition. This translation process would have to be undertaken for each specified condition for the corresponding indicator. In cases where the policy defines one benchmark condition as a status relative to another condition in the policy, later iterations may have to use the results of a previous iteration.
- Often a single policy will be implemented using several indicators. The ecological reasoning used for setting reference levels for each condition should be as consistent as possible across the indicators, to ensure the indicators are used as consistently as possible in evaluating ecosystem status relative to the policy objectives.
- The operational complications in Section 5.3 are relevant here as well.
- Sometimes an indicator may be relevant to several policies, and the status required for compliance with the different policies may not be the same (some aspects of the WFD and MSFD, for example). How this potential contradiction should be resolved depends on how the policy interacts legally.
- There is no guarantee that all science experts will agree on all aspects of interpreting and applying the science knowledge. This is not a problem unique to indicators, and Sections 6.4 and 6.5 includes some guidance on this issue.
- Science answers are always uncertainty to various degrees, and there are benefits form doing all this work in risk-based frameworks, as well as often additional demands on science to apply those frameworks.

As reviewed in Section 6.2.3.2, the MSFD has policy objectives for both prosperity through sustainable use, and for conservation of ecosystem structure, function and processes. The ICES/JRC Task Group Reports and the Elements of a Decision which followed from those reports has a number of candidate indicators that are more naturally associated with sustainable use. Indicators associated with the Descriptors Food Webs and Commercial Fish are possibly the clearest examples of this type of indicator. However some candidate indicators are more naturally associated with striving for unimpacted conditions, particularly those associated with the Descriptor on Contaminants and hazardous substances.

WGECO encourages that Section 4 of this report be considered by all groups working on setting reference levels that reflect sustainable use. In the remainder of this Section we provide the corresponding guidance on ecological reasoning and steps for implementation, when ecologically consistent reference levels are needed for an unimpacted state of an indicator. This is intended to be *incremental to* the information in Section 4 on sustainability and not a replacement for any of it.

Pristine:

To set a reference level for pristine conditions, it is necessary to have some idea of what state the indicator would have been in, at a time where human activities were not impacting the parts of the ecosystem measured by the indicator. Sometimes this can be done directly, when the indicator measures an ecosystem attribute like a contaminant that did not exist in the waters before humans introduced it to the system. If this condition is met, then the reference level is absence of the ecosystem attribute being measured.

If the indicator is measuring an attribute that always existed in the ecosystem but may have its status altered by human use, then it is necessary to have some way to estimate what its status was before human activities began to impact the attribute. If a data-series is sufficiently long, it may be possible to find a reference period that represents the value of the indicator prior to human impacts. That condition will rarely be met, but when it is, the value of the reference level on the indicator should be the value of the indicator in the unimpacted period. In some cases, where ecosystem attributes are relatively stationary in space, it may be appropriate to use reference areas where the activity has not occurred, but this can only be considered if the reference areas have the same environmental features and physico-chemical attributes as the impacted areas.

If there are no direct measures of the status of the indicator in a time (or area) before human impacts, then some scientifically sound method will be required to project backwards what value the indicator would have had prior to human impacts. Such projections are always uncertain, these uncertainties need to be considered and communicated clearly whenever a reference level is chosen by this method and then used in monitoring and assessment. If there is sufficient knowledge that a process-based model of how the attribute measured by the indicator varies due to natural processes and is affected by pressures associated with human uses, then the model can be used to estimate the value the attribute measured by the indicator would have (or range in which it would vary) in the absence of those human-induced pressures. The reference level for the indicator would then be the value the indicator would be expected to take on, when the ecosystem attribute was in the predicted range.

If there is not sufficient knowledge for a process-based model, then monitoring data for the indicator over a period when both the ecosystem attribute and the pressures associated with human use were varying can be used in statistical models to estimate the value the indicator would have taken on when the pressures were zero. The uncertainty of these statistical forecasts will depend on the contrast in available data to parameterize the statistical model, and the degree to which the functional relationship between the indicator and effects of the pressures can be specified (see Section 4.3), and can be quite large.

As a last resort, expert opinion may have to be used to provide an informed opinion at the value the indicator would take on, were the system was not impacted by human uses that affect the attributes being measured by the indicator.

Complications:

- Taking account of the sensitivity of the measurement process and sampling design producing the indicator.
- For the "unimpacted time" (or area) standard, dealing with natural variability in the ecosystem attribute during that time (or across and within areas).
- Taking account of natural changes that the ecosystem might have undergone such that even if human uses had not altered the ecosystem, the value of the indicator would not be the same now as it would have been in some earlier period when humans had not altered the ecosystem property.
- In some cases the value of an indicator and the value of the ecosystem attribute it measures are identical, such as when the indicator is a density or a rate. However, often the indicator is only an index of the ecosystem attribute, and some scaling of the indicator value to the ecosystem attribute is needed. This needs to be taken into account when using process-based

or statistical models to estimate an unimpacted reference level. Note that this scaling is not introduced by this approach to setting reference levels; it is necessary any time the indicator is an index and not a direct measure of the ecosystem attribute of interest.

6.3.3 How to combine information across indicators for an overall assessment of "good environmental status"

6.3.3.1 Considerations and framework for combining indicators within the MSFD

The assessments of GES at regional and subregional scales will be based on the indicators that have been chosen and the reference level set for them. The reporting on GES will require not just determining that status of ecosystems on the individual indicators relative to their reference levels, but also combining that level across indicators into integrated conclusions and statements about GES. How this combining of information is done may be as important to the assessments as the choices of indicators and decisions about reference levels. The assessments of GES need to convey clearly both the better and worse aspects of environmental status in each region and the progress made from one assessment to the other. Poor choices of methods for combining information could have several consequences. As described in 6.3.3.2 inappropriate methods for combining across indicators could obscure either successes or failures at achieving GES in subareas of the reporting region or across differing parts of the whole ecosystem. Poor choices could also make it hard to evaluate progress from one assessment to the next, or the effectiveness of the set of measures that have been adopted for addressing shortcomings in GES identified in past assessments.

The EU, Member States, and those associated with developing the assessment framework (in particular the ICES-JRC Task Groups and WG GES) are well aware of the importance of integrating across indicators. All the task groups provide views on how it should be done with their individual reports. The preferred approaches differ among groups, reflecting in substantial part the different histories of applied work in Europe on different ecological topics which is rooted in the Water Framework Directive for some Descriptors, the Habitats Directive for others, and for still others policies like the Common Fisheries Policy. As reviewed in 6.2, these Directives differ in how status relative to objectives is assessed. The conclusions of each Task Group are summarized briefly in Table 1 from the Management Group Report, extracted here as Table 6.3.3.1.1.

The Management Group Report also includes specific guidance on the combination of information across indicators. It is the Management Group Report which considers the issue in the full context of evaluating GES across Descriptors and on larger scales. Building on work of one of the Task Groups it also contains a fuller discussion of the possible strategies for combining information across indicators, and implications of various options.

Table 6.3.3.1.1. Table 1 from the Management Group Report Describing Integration of Descriptors.

Descriptor	Integration
Biodiversity	Integrative assessments combining indicators and reference levels appropriate to local conditions
NIS	Integrative assessments combining indicators and reference levels appropriate to local conditions
Commercial fish	3 attributes, descriptor not ok if any attribute is not ok
Food webs	2 attributes, descriptor not ok if any attribute is not ok
Eutrophication	4 attributes. Integrative assessments combining indicators and reference levels appropriate to local conditions
Sea floor	Integrative assessments combining indicators and reference levels appropriate to local conditions
(Morphology)	[Not available at time of Management Group meeting]
Contaminants	3 attributes, descriptor not ok if any attribute is not ok
Food contaminants	1 attribute (below regulatory limits)
Litter	3 attributes, descriptor not ok if any attribute is not ok
Noise	3 attributes (1 indicator for each), descriptor not ok if any attribute is not ok

The Management Group report summarizes the challenge to combining information as:

"There are two or three levels of integration required to move from evaluation of the individual indicators identified by the Task Groups to an assessment of Good Environmental Status (GES);

- *Indicators within individual Attributes of a Descriptor (for complex Descriptors)*
- *Status across all the Attributes within a Descriptor*
- *Status across all Descriptors*

As one moves up these scales the diversity of features that have to be integrated increases rapidly. This poses several challenges arising from the diversity of units, scales, performance features (sensitivity, specificity, etc) and inherent nature (state indicators, pressure indicators, response indictors) of the measures that must be integrated.

The evaluation of GES will have to balance two undesirable but inescapable compromises; having an evaluation methodology that is scientifically sound and makes best use of available information, and having an evaluation methodology that is consistent in all applications – consistent with regard to the types of information used and the methods applied in their use. Increasing consistency in methods at regional and large sub-regional scales will come at a cost of requiring use of suboptimal and sometimes inappropriate indicators, benchmarks, and analytical algorithms. Increasing the matching of methods to specific conditions within each regional sea (or sub-regional sea) will come at a cost of less consistency in practice within the larger scales."

The Report then discusses how suitable practices for combining information across indicators might differ between assessments of GES at local scales. It notes that even at local scales:

"The evaluation [of GES] should not focus on providing a single number for the local area, particularly if the area is chosen to reflect a known pressure gradient. Rather it should integrate the information in the suite of indicators and benchmarks into a clear, concise, but

multi-factorial reflection of the status of e.g. the seafloor community within the locale or along the pressure gradient. However, the evaluation might wish to achieve this through a relatively fully specified algorithm for using the set of indicators and benchmarks. Such algorithms can only be developed and parameterized on the scale at which they will be used. No universal algorithms exist. … At larger scales … it is neither feasible nor ecologically appropriate to specify prescriptive lists of indicators and analytical algorithms for evaluating GES. Too many compromises would have to be made in choosing indicators that were robust but could not make full use of available and relevant information and in assigning compromise weightings and benchmarks that were likely to be suboptimal in each contributing area. More importantly, there would be a merging and likely obscuring of much information important for understanding where the successes and failures in progressing towards GES were occurring, and in informing decision-makers about where policies and management were working well and where adaptation or innovation in policy and manager were needed."

WGECO agrees that often at local scales, and particularly regional and subregional scales, evaluations of GES should not be based on approaches which simply roll up status across all indicators into a single number which is used as an index or "Score" for GES. This includes binary approaches such as a "one out – all out" algorithm where failure to be meet or exceed a reference level on one indicator means a reporting of "Failure" for GES. Such approaches fail to reflect the information available in the assessment of GES in ways that are most helpful to the public, who will be engaged in a dialogue about the outcome of the assessment, decision-makers who will have to choose what measures to add or adapt after each assessment, or the scientific and technical experts who will have to advise both the public dialogue and the policy choices.

The Management Group Report concludes that:

"What is needed for combining the information available on the diverse attributes of e.g. seafloor integrity is not some fully specified and well-structured analytical method for assessing GES, but a fully specified and well-structured process for conducting assessments of GES. [such a process] will provide the only realistic avenue for having regular evaluations of GES at regional and large sub-regional scales. The periodic (possibly, but not necessarily, annual) assessments would not have a single framework or template that would be the required approach. Rather the process could adapt practice from assessment to assessment with regard to indicators selected, weightings and benchmarks applied, and approaches to integrating local scale evaluations into regional conclusions based on the developing experience and knowledge."

This conclusion highlights the key points that are needed for combining information across indicators in assessments generally and in particularly for assessments of GES. The methods need to be flexible rather than prescriptive; clarify rather than obscure differences in status among indicators and among subareas within the larger area being assessed; interpretable relative to common standards but not forced to apply identical benchmarks. However the need for the assessment process to be flexible and adaptive should not be license for *ad hoc* approaches to assessment of GES. Approaches for combining information across indicators should be based on the strength of the methods to reveal patterns across indicators and space and definitely not obscure them, to communicate uncertainty rather than cover it up, and to guide follow-up dialogue to focus on what is working well and where problems should be addressed rather than on merely whether the ecosystem "passed" or "failed" the assessment. The options for combining results across indicators in Section 6.3.3.2 should be considered in this context. In addition, based on experience in processes for producing sound science products on complex issues, WGECO also considered some of the aspects of the process for the assessment in Section 6.4.

In fact, looking back at the Management Group conclusion that a process for assessment "*will provide the only realistic avenue for having regular evaluations of GES*", one may ask how much guidance might be provided in future to the operation of those processes. As reviewed in the TG 6 Report, attempts to require a single analytical assessment method and common tuning indices for all fish stocks have quickly been abandoned because they prevent the use of new information and provide suboptimal results for all but the most "typical cases". Likewise there is little reason to support the highly formulaic approach of some indices used in the WFD, for the more complex assessments of GES. Nonetheless, after working at the problem for only a few years, WGECO is now able to provide substantial guidance on setting ecologically consistent reference levels for diverse types of indicators (Sections 4 and 6.3.2). It is possible that should equal effort to be spent adapting the experiences with integrate ecosystem assessment methods (Section 3) and other multi-indicator methods of evaluating ecosystem status and trends, comparable guidance may be possible for the assessment process as well.

6.3.3.2 Methodological options

To assess ecosystem status, an integrative approach should focus on indicators of ecosystem health, such as vigour (productivity), organization (diversity, connectivity), and resilience (Rapport *et al.*, 1998) while an analytical approach following the MSFD, will aim at assessing each system component separately. Several analytical approaches have been tested for marine ecosystems. Most attempts include some concept of pressure and impact, either in matrix form (ICES, 2007) or in a more discursive way (Link and Brodziak, 2002; DFO, 2003). When it comes to combining assessments for a wide diversity of aspects (pressures or impacts, or both), several families of approaches have been suggested. The first one consists in a simple graphical representation, for example radar plots have been proposed for display of ecosystem status, using semi-quantitative information on a selection of ecosystem indicators (Collie *et al.*, 2003). The second consists in combining indicator values into a single number by averaging or scoring: impacts are scored and scores are combined into a composite index. One of the most developed examples of this approach is the Integrated Biotic Index (IBI) used for assessing ecological integrity of rivers (Fausch *et al.*, 1984). Scoring methods are generally developed *ad hoc* and may, or may not, include weighting of separate components (ICES, 2007; Halpern *et al.*, 2008). The third type of approach requires time-series data for a wide diversity of components and assesses their common dynamics using multivariate analyses. The results can be presented in 'traffic light' tables (DFO, 2003; Caddy and Surette, 2005; ICES, 2006). The last family of approaches is in development and relies on the use of a model that assumes relationships between indicators and can be utilised to estimate the parameters of these relationships (rates, slopes, strengths) or to test for their significance. Models might be quantitative such as the ones to be used in the US approach to IEA (Levin *et al.*, 2009), or qualitative (see below).

All four families of analyses will generally involve implicit or explicit standardization, weighting and combination of steps. Aggregation methods have a risk of concealing the nature of what is being perturbed. Moreover, state indicators pointing to one direction, might conceal others trending in the opposite direction (the ''eclipse'' effect). Weighting could be used for methodological reasons (redundancy, unequal uncertainty among indicators), or for policy reasons. This makes changes in the weighted value of the aggregate score difficult to interpret without returning to the patterns observed in the individual indicators. Strengths and weaknesses of most of these methods have been reviewed in Rice and Rochet (2005) and are summarized in Table 6.3.3.2.1.

The assessments needed for implementing the MSFD are expected to include an explicit description of the relationships between pressure and state, and a common approach that will ensure consistency and comparability across marine regions (see Section 3.3 of this Report). Multiple impacts and socio-economic aspects should also be covered (MSFD Article 8). Most of the existing methods do not explicitly address these characteristics (Table 6.3.3.2.1). Another issue lies in the holistic nature of the MSFD requirements, which means that the best information available on some components will be very limited or even missing. Thus any integration method should acknowledge this uncertainty and make a due treatment of knowledge gaps. Therefore below we introduce three methods in development that address i) multiple impacts (Section 6.3.3.2.1) ii) pressure-state links and multiple impacts (Section 6.3.3.2.2) and iii) acknowledgement of uncertainty (Section 6.3.3.2.3).

Table 6.3.3.2.1. Strengths and weaknesses of various integration methods, with a focus on characteristics particularly relevant to the MSFD (see Section 3.3): Pressure-State link (P-S link), and consistency and comparability (C-C).

METHOD	STRENGTHS	WEAKNESSES	P-S LINK	C-C
Kites, pie diagrams	Simple and transparent	Does not accommodate large indicator suites; ± prone to manipulation	No	No
Averages	Simple and understandable	Eclipsing; weighting issues	No	No
Composite indices		Eclipsing; not transparent	No	Yes
Multivariate methods (canonival correlation…)	Reduces redundancy, accounts for uncertainty	Data greedy	Exploratory and correlative	Probably not
Traffic light presentation	Easy to understand	Data greedy; weighting issues	Visual	No
Model-based methods (in development)	Rely on a consistent set of interrelated indicators	Complex models not transparent; complex output may need summarizing	Explicity assumed	Potentially

6.3.3.2.1 Multiple pressures and impacts

The risk and consequences of aggregate impacts on ecosystems due to the presence of multiple pressures is discussed in a number of theoretical and practical ecological studies but still its quantification is a difficult task. Recent studies to explore the nature of interaction of multiple pressures show that aggregate impacts are additive (i.e., are summed) for pairs of pressures (Crain *et al*., 2008) and that synergistic effects are generally more common than additive ones (Darling and Côté, 2008). This highlights the need to evaluate the complexity and range of uncertainty in assessing aggregate (and cumulative) impacts of human pressures.

Assuming that multiple activities act independently within a system recent studies by Ban and Alder (2008) and Halpern *et al*. (2008) modelled cumulative impacts as the additive accumulation of impacts of individual activities combining a measure of ecosystem component sensitivity and the risk of occurrence of an activity. In contrast, Stelzenmüller *et al*. (2010) used generic pressure categories exerted by human activities and developed a range of models that quantify the risk of cumulative and/or aggregate impacts on marine habitats. More precisely, their geospatial modelling framework used the footprint and intensity of a number of human pressures, measures of habitat sensitivities to those pressures and a process that allowed the alteration of the importance of single pressures. This resulted in a number of scenarios for risk of aggregate impacts with numerical results other than the addition of single pressures. This framework shows a high level of flexibility as it can be applied at any spatial scale and adapted to different pressure categories when suitable data are available. Moreover, depending on the spatial scale of its application it can be modified to focus on multiple activities rather than on multiple pressures by omitting some steps.

The aggregate impact of multiple activities was recently modelled at the scale of ecoregions and benthic habitats for Canada's Pacific area assuming a linear decay from the origin of the activities (Ban *et al*., 2010). The authors considered specifically deep pelagic waters and shallow pelagic waters and give therefore another example on how different scenarios for the risk of aggregate impacts can be developed. Common limitations to all of the above listed studies are the lack of experimentally assessed information on the sensitivity of ecosystem components and a more comprehensive knowledge on the interactions of human activities. Especially when more information on the latter becomes available current modelling approaches can be developed further to assess the risk of cumulative and aggregate impacts also on the basis of synergistic and antagonistic effects of multiple human activities.

6.3.3.2.2 Combining trends

Averages and composite indices provide little understanding of what is actually changing and why. Moreover, divergent trends in component-metrics might cancel each other and precious information on metric-specific sensitivities is lost. There have been, however, attempts to identify causes of changes by combining changes in indicator species that react differently to different sources of pollution (Lenihan *et al*., 2003), or in population metrics more sensitive to variations in mortality or recruitment (Trenkel *et al*., 2007). This approach has been generalized to combining trends in multiple metrics in order to detect the effects of changes in major pressures, and help identify likely causes of impacts.

First, predictions of expected changes in state given changes in pressures are obtained by qualitative analysis, that is, a mathematical analysis of direction, not amount, of changes (Dambacher *et al*., 2009). Second, metrics that describe changes in the model variables are selected. Third, a method to combine trends in metrics, while taking account of uncertainty and variability has been developed to identify for which causes of impacts there is evidence in the data. The likelihood principle is used to select causes that best explain observed trends in metrics, and log-likelihood values are summed to combine evidence across metrics, populations, and organization levels (Trenkel and Rochet, in press). In a comparative study across 14 exploited shelf groundfish communities, this method proved powerful in detecting changes and identifying their likely causes (Rochet *et al*., submitted). Generally several impacts were found likely, generating ambiguous results. This is partly inherent to using noisy data and indirect evidence, a constraint unavoidable when more complex assessments need to be carried out, accounting for multiple pressures and interactions. Ambiguous results are a way to acknowledge uncertainty. This could be used in a precautionary manner in subsequent management decisions: if among several identified causes one is manageable, precaution would lead to act on this cause to reverse the trends. This method for synthesizing information across ecosystem components and data sources has potentially wide applicability as pressure and/or socio-economic metrics could be incorporated as well. The approach mainly focuses on changes and linkages and consequently does not require that reference levels are established for any indicator.

6.3.3.2.3 Fuzzy set method

This method pertains to the averaging family, but is being developed to take account of uncertainty. In most applications of the analytical approach, little attention has been paid to the lack or poor quality of information for some ecosystem components. Halpern *et al*. (2007), in a survey of marine ecosystem vulnerability to anthropogenic threats, asked experts to score the certainty of their evaluation from none to very high (extensive empirical or personal knowledge), and then used these certainty values to weight scores across experts. This is a first step towards accounting for uncertainty, but it is limited to scoring results, and does not account for complete ignorance. The important question is therefore how to keep track of uncertainty across all components and how to include components with no information at all.

The fuzzy set approach (Zimmermann, 2001) has been suggested by Silvert, 1997; 2000 for including uncertainty in ecological assessments. Fuzzy set elements have partial membership that can be determined based on a combination of assessment criteria and certainty of knowledge. For example, a fish stock reliably assessed to have spawning stock biomass above a certain minimum reference point has a good membership to the set 'GES', while a species that has not been observed in the wild in recent years has a low membership to this set. Non-assessed species would be assigned an intermediate membership to acknowledge ignorance. Fuzzy set membership can be combined across elements and, if sets are taken as elements of a wider set, for example say the set of impacted ecosystem components, the results can be combined across sets. Thus, the fuzzy set approach offers a way to incorporate uncertainty when combining a diversity of specific assessments.

Fuzzy set methods admit a diversity of rules to combine the memberships of several indicators to obtain the membership of a given qualitative descriptor to the set of descriptors with good environmental status and subsequently of all descriptors for the ecosystem level assessment. Combination rules can be non-compensatory, in which case a low membership of one indicator does not compensate for a high membership of another indicator, balanced, high and low memberships balance each other, or unbalanced, giving more weight to low memberships. The minimum is a typical non-compensatory combination rule (membership of a group is the minimum membership of its members); the arithmetic mean is a balanced rule; the geometric mean, or other parameterized combination rules can be used if unbalanced rules are desired. The choice of a combination rule is up to the users of the assessment, depending on the kind of assessment they wish. Among the GES descriptors of the European Marine Strategy, descriptor QD3 'Commercial fish' implies a non-compensatory combination rule: "Populations of all commercially exploited fish and shellfish are within safe biological limits"; the conditions for other descriptors are less explicit. According to Table 6.3.3 above four other descriptors should use a non-compensatory rule: food webs, contaminants, litter, and noise. To avoid eclipsing it is essential that any combined GES membership be accompanied with the detailed membership of less aggregated levels, for example the overall status should be presented together with the assessment for separate descriptors, and assessment for each descriptor should in turn be associated with a summary of assessments for each attribute (example: Figure 6.3.3.2.3.1, cf. Rochet *et al*., in preparation).

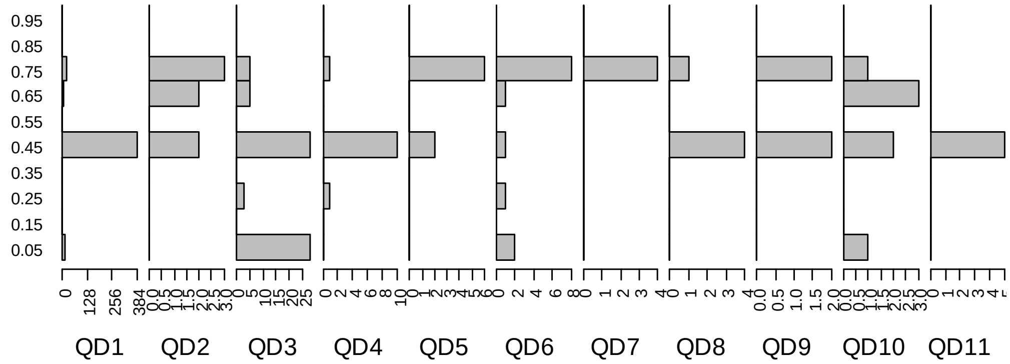


Figure 6.3.3.2.3.1. Histograms of partial membership to 'GES' for each qualitative descriptor based on data with contrasted uncertainty across descriptors and attributes. The y-axis reflects membership to GES: a value of 1 would be certain, 0 would be non-GES certain, and 0.5 implies ignorance of actual status. Within each descriptor the bars present the distribution of attributes along this gradient.

6.4 Processes for the next step

6.4.1 Considerations from assessment of assessments

Last year WGECO (ICES, 2009) considered the "design features" and "best practices" for assessment processes that were recommended by the Group of Experts for the Assessment of Assessments (AofA; IOC-UNESCO, 2009). They were considered generally appropriate as a framework for ecological assessment processes, and would provide a useful basis for designing the assessment processes for evaluating GES. The key design features (and their descriptions) in the AofA report are listed below, along with brief commentaries on their relevance to assessments of GES within the MSFD:

a) **Objectives and Scope:** clear goals and definitions; progress toward integrated marine assessment and ecosystem approaches and progress toward regular, iterative assessment in support of adaptive management that links potential solutions to identified problems; Commentary: These are set by the MSFD and Elements of the Decision.

b) **The Science/Policy Relationship**: regular dialogue, policy-relevant questions, guidance for priority-setting, identified target audience(s) and the roles of governments and other stakeholders vis-à-vis experts, including government involvement in reviewing assessment products; Commentary: There is no guidance in the MSFD on this. However, the relationship is particularly important for positioning the assessments of GES in the overall implementation of the Directive.

c) **Stakeholder Participation:** clear and meaningful modalities for participation by stakeholders; Commentary: There is no guidance in the MSFD on *how* this is to be achieved, but both the Directive and the processes used by EC for progress towards implementation are creating high expectations among stakeholders for engagement in all aspects of implementations. These expectations will require effective management.

d) **Nomination and Selection of Experts**: transparent criteria and procedures for selecting lead authors, contributing authors, peer reviewers and other experts; provision for balance and to protect the integrity of the process from inappropriate influence and bias (e.g., from employers, funders or sponsoring bodies); Commentary: This task has been assigned to States. It will be a key step, to ensure appropriate breadth of knowledge and balanced perspective by the assessment teams.

e) **Data and Information**: agreed procedures for sourcing, quality assurance and the availability and accessibility of underlying data and information including metadata; clear standards for reporting on the extent, representativeness and timeliness of available data and the presence of any significant gaps; methods for scaling information up or down and on methods for drawing inferences to reach general conclusions including implications for assessment findings; Commentary: This will strongly influence the selection of indicators for use in each regional or national assessment, but once the indicators are chosen should only require revisiting periodically.

f) **Treatment of Lack of Consensus among Experts**: clear and transparent guidelines for addressing and reporting lack of consensus; Commentary: Relevant but no unique considerations relative to MSFD.

g) **Treatment of Uncertainty**: clear and transparent guidelines for addressing and reporting uncertainty; Commentary: Important but no unique considerations relative to MSFD.

h) **Peer Review:** agreed, transparent criteria and procedures; use of reviewers not involved in the assessment; Commentary: Very important and discussed in 6.3.4.2.

i) **Effective Communication**: provision to develop a communications and outreach strategy to cover the entire period of the assessment, including appropriate products for each identified target audience; Commentary: Important and little evidence yet of planning for this task.

j) **Capacity-Building and Networking**: strategies for improving assessments over time through targeted efforts; Commentary: Networking of those engaged in assessments across Europe will be crucial to success, and great opportunities for capacity building are present.

k) **Post-Assessment Evaluation**: provision for post-assessment evaluation of assessment products and the assessment process itself, drawing both on *insiders* involved in the process and *outsiders* not involved in the assessment in any way; Commentary: Given the role of these assessments in policy and management in Europe it is expected that these assessments will receive lots of post-evaluation, from many quarters.

l) **Institutional Arrangements**: clear agreement on the composition of institutional mechanisms and relationships between them; clearly articulated responsibilities for management and expert components and for the secretariat; development of a networked "system" of assessment processes. Commentary: High-level arrangements are largely specified by the MSFD, much detail remains to be worked out at the implementation level. States are actively working on the institutional arrangements for implementation.

6.4.2 Assessment process issues and the MSFD

Section 6.4.1 considered briefly the conclusions on design features and best practices for assessment processes from the UNEP/IOC Assessment of Assessments (AofA). Although the basis for the AofA conclusions was social science research in which WGECO does not have particular expertise, the conclusions are consistent with the experience of WGECO as part of the ICES processes for assessments and provision of science advice. We agree these features and practices should be important guidance in developing the processes at the national (or possibly in future regional) level which will actually do the assessments of GES. We provide several suggestions for those processes might best operate, if the products (the national assessments of GES) are to be scientifically sound, and be legitimate, credible, and relevant. We also stress that the public and institutional policy discussions that will follow from release of the periodic assessments will need to be informed by another level of science-based process, which will have several important roles outlined below.

With regard to the processes used for the GES assessments:

The processes should be expert processes following best scientific practices to get the "right" answer. They should not function like consultation processes, intended to produce "the most popular" answer. This implies the need for *a priori* standards to be set for membership on the assessment teams; standards that would allow experts in both traditional/experiential knowledge and the natural and social sciences. It also implies that process of the assessment would not be exposed to partisan pressure from any direction while the assessments were under way, whether the pressure might come from the political arena, public advocacy groups, or even other science experts with narrowly focused interests or strongly-held interpretational perspectives. It is the governance process which manages risks, *informed by* the assessments. The experts doing the assessments should not imbed their own risk tolerances in the assessments. Finally, it implies the need for an approach which is, to the extent possible, evidence based and includes provisions for transparency and peer review.

Because of the flexibility expected in choices of indicators and reference levels, if the proposals in Section 6.3 are generally followed, it may be particular challenging to have scientifically sound evaluations of GES in the first few assessments. The following steps might be helpful in addressing that challenge.

Once the decisions have been made about the indicators that will be used in each national or regional assessment, the expert assessment processes should immediately undertake some specific tasks. First, they should identify the reference levels for each indicator, well before the first assessment that will use them is needed. This increases the likelihood that the reference levels would be determined by the knowledge of the ecosystems, and not the outcomes that they will produce. Second, once the reference levels are set, a hypothetical "good GES case" assessment could be conducted. The core "hypothetical assessment" would assume all indicators were at their reference levels, with a few alternative scenarios hypothesising perhaps minor shortcomings in many areas or major problems with a few pressures. It would serve several functions. The usefulness of different methods for combining information across indicators could be explored, to see how effective they were at identifying problems that were known to be present, and communicating that information in their products. An important function of the core hypothetical assessment would be its representation, both in analytical results and particular in their narrative interpretation, of what the region would be like, if GES actually were achieved. Knowing what GES would actually look like and how it would be described within the framework and reference levels that had been adopted could be an invaluable benchmark for all future assessments. Depending on the assessment methods used, and the ability to work from desired value of a state indicator to necessary value if a pressure indicator (see Section 4.3) it might even be possible to determine at what levels the dominant pressures in the region would have to be for the indicators to all be at their reference levels. This could support an initial dialogue on not just what activities will have to be managed to achieve GES, but how stringent the management will have to be.

The MSFD does not call for any "meta-process" above the GES assessments, but we see several important functions which require one. All are related to the need for conclusions to be drawn about "consistency" across assessments that are done independently, with flexibility to choose indicators, set reference levels, and apply methods of analysis and integration of information, as well as real differences among the ecosystems. There is a strong science component to decisions about whether sets of indicators are "functionally equivalent" (Section 3), and whether they cover the important properties of ecosystem structure and function, the dominant pressures and the important interactions in an adequately complete and balanced way. There is a strong science component to deciding if reference levels are also "functionally equivalent" even if they differ in absolute value (Sections 4 and 6.3). Several multi-institutional projects are investigating aspects of the frameworks for assessments in a research context, but there are important differences between research investigations and arm's length peer review and provision of advice. An arms-length science advisory body working on a European scale could serve well to fill those science needs. Were such a body given the role of evaluating both adequacy and "consistency" of indicator choices and reference levels, it also could serve the important function of independent peer review of the individual assessments themselves. Such a process would be able to make a particularly valuable contribution at the start of this major cycle of GES assessments, by evaluating the "consistency" of the hypothetical "good GES" assessments suggested in the preceding paragraph. If these were found not to be consistent in their interpretation of GES, the problems could be diagnosed and addressed before they might become major policy hurdles in developing action within the EU on the real assessments that would soon follow.

6.5 References

Ban, N., Alder, J. 2008. How wild is the ocean? Assessing the intensity of anthropogenic marine activities in British Columbia, Canada. Aquatic Conservation: Marine and Freshwater Ecosystems 18, 55–85.

Ban, N.C., Alidina, H.M., Ardron, J.A. 2010. Cumulative impact mapping: Advances, relevance and limitations to marine management and conservation, using Canada's Pacific waters as a case study. Marine Policy, doi:10.1016/j.marpol.2010.01.010.

Caddy J. F. and Surette T. 2005. In retrospect the assumption of sustainability for Atlantic fisheries has proved an illusion. Reviews in Fish Biology and Fisheries 15: 313–337.

Cardoso, A.,C, Cochrane, S., Doerner H., Ferreira, J.G., Galgani, F., Hagebro, C., Hanke, G., Hoepffner, N., Keizer, P.D., Law, R., Olenon, S., Piet, G.J., Rice, J., Rogers, S.I., Swartenbroux, F, Tasker, M., and van de Bund, W. 2010. Marine Strategy Framework Directive, Management Group Report. Luxembourg Office for Official Publications of the European Communities, 59 pp.

Collie JS, Gislason H, Vinther, M. 2003. Using AMOEBAs to display multispecies, multifleet fisheries advice. ICES J. Mar. Sci. 60: 709–720.

Crain, C.M., Kroeker, K., Halpern, B.S. 2008. Interactive and cumulative effects of multiple human stressors in marine systems. Ecology Letters 11, 1304–1315.

Dambacher J. M., Gaughan D. J., Rochet M. J., Rossignol P. A. and Trenkel V. M. 2009. Qualitative Modelling and Indicators of Exploited Ecosystems. Fish and Fisheries 10: 305–322.

Darling ES, Côté IM (2008) Quantifying the evidence for ecological synergies. Ecology Letters 11, 1278–1286

DFO. 2003. State of the Eastern Scotian Shelf Ecosystem. DFO Can. Sci. Advis. Sec. Ecosystem Status Rep. 2003/004 25 pp.

European Commission. 2001 Assessment of plans and projects significantly affecting Natura 2000 sites. Methodological guidance on the provisions of Article 6(3) and (4) of the Habitats Directive 92/43/EEC.

European Commission. 2009 Composite Report on the Conservation Status of Habitat types and Species as required under Article 17 of the Habitats Directive (Report from the Commission to the Council and the European parliament) COM(2009) 358.

European Commission. 2010. Elements for the Commission decision on criteria on good environmental status under article 9(3) MSFD. 17 pp.

Fausch K. D., Karr J. R., and Yant P. R. 1984. Regional application of an index of biotic integrity based on stream fish communities. Transactions of the American Fisheries Society, 113: 39–55.

Halpern B. S., Walbridge S., Selkoe K. A., Kappel C. V., Micheli F., D'Agrosa C., Bruno J. F., Casey K. S., Ebert C., Fox H. E., Fujita R., Heinemann D., Lenihan H. S., Madin E. M. P., Perry M. T., Selig E. R., Spalding M., Steneck R., and Watson R. 2008. A global map of human impact on marine ecosystems. Science 319: 948–952.

Halpern B. S., Selkoe K. A., Micheli F., and Kappel C. V. 2007. Evaluating and ranking the vulnerability of global marine ecosystems to anthropogenic threats. Conservation biology 21: 1301–1315.

ICES. 2006. Report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO). WGECO Report 2006 ICES Advisory Committee on Ecosystems, ACE:05, 179pp.

ICES. 2007. Report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO). ICES Document CM 2007/ACE:04. 162pp.

ICES. 2009. Report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO). WGECO Report 2009 ICES Advisory Committee on Ecosystems, ACE:20, 190pp.

JNCC. 2007. Second Report by the UK under Article 17 on the implementation of the Habitats. Directive from January 2001–December 2006. Peterborough: Joint Nature Conservation Committee.

Lenihan H. S., Peterson C. H., Kim S. L., Conlan K. E., Fairey R., McDonald C., Grabowski J. H. and Oliver J. S. 2003. Variations in marine benthic community composition allows discrimination of multiple stressors. MEPS 261: 63–73.

Levin, P.S., Fogarty, M.J., Murawski, S.A., Fluharty, D. 2009. Integrated ecosystem assessments: developing the scientific basis for ecosystem-based management of the ocean. PloS Biology 7: 23–28.

Link J. S., and Brodziak J. K. T. eds. 2002. Status of the Northeast US continental shelf ecosystem. Northeast Fisheries Science Center Reference Document 02-11 Northeast Fisheries Science Center Woods Hole Massachussetts. 245 pp.

Rapport D. J., Costanza R. and McMichael, A.J. 1998. Assessing ecosystem health. Trends in Ecology and Evolution 13: 397–402.

Rice J., Rochet M.J. 2005. A framework for selecting a suite of indicators for fisheries management. ICES J. Mar. Sci. 62: 516–527.

Rochet M.J., Trenkel V.M. 2009. Why and how could indicators be used in an ecosystem approach to fisheries management? In: Future of Fishery Science in North America. Beamish R.J., Rothschild B. (eds). Springer. pp. 209–226.

Rochet MJ., Trenkel VM., Lorance P. In preparation. Accounting for uncertainty when evaluating the status of marine ecosystems: application to the Bay of Biscay.

Rochet MJ., Trenkel V., Carpentier A., Coppin F., Gil de Sola L., Léauté JP., Mahé JC., Maiorano P., Mannini A., Murenu M., Piet G., Politou CY., Reale B., Specidato MT., M.T., Tserpes, G., and Bertrand, J.A. Submitted. Do changes in environmental pressures impact marine communities? An empirical assessment.

Silvert W. 1997. Ecological impact classification with fuzzy sets. Ecological Modelling 96: 1–10.

Silvert W. 2000. Fuzzy indices of environmental conditions. Ecological Modelling 130, 111–119.

Stelzenmüller, V., Lee, J., South, A., Rogers, S.I. 2010. Quantifying cumulative impacts of human pressures on the marine environment: A geospatial modelling framework Marine Ecology Progress Series 398, 19–32.

Trenkel V. M., Rochet M.-J. In press. Combining time trends in multiple metrics for identifying persistent changes in population processes or environmental stressors. Journal of applied ecology.

Trenkel V. M., Rochet M.-J., Mesnil B. 2007. From model-based prescriptive advice to indicator-based interactive advice. ICES Journal of Marine Science 64: 768–774.

Tserpes G., Bertrand J. Submitted. Do changes in environmental and fishing pressures impact marine communities? An empirical assessment.

UNEP and IOC-UNESCO. 2009. An assessment of assessments, findings of the group of experts. Start-up phase of regular process for global reporting and assessment of the state of the marine environment including socio-economic aspects. ISBN 978-92-807-2976-4, United

Nations Environment Programme and the Intergovernmental Oceanogrphic Commission of UNESCO, Valetta, Malta.

Zimmermann H.-J. 2001. Fuzzy set theory - and its applications. Boston: Kluwer Academic Publishers.

Zingstra H., Kovachev A., Kitnaes K., Tzonev R., Dimova D., Tzvetkov P. 2009. Guidelines for assessing Favourable Conservation Status of Natura 2000 Species and Habitat Types in Bulgaria. Executive Summary. Bulgarian Biodiversity Foundation, Sofia (English).

2005/646/EC. COMMISSION DECISION of 17 August 2005 on the establishment of a register of sites to form the intercalibration network in accordance with Directive 2000/60/EC of the European Parliament and of the Council.

2000/60/EC. DIRECTIVE OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 23 October 2000 establishing a framework for Community action in the field of water policy.

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

2008/915/CE. Commission Decision of 30 October 2008 establishing, pursuant to Directive 2000/60/EC of the European Parliament and of the Council, the values of the Member State monitoring system classifications as a result of the intercalibration exercise (notified under document number C(2008) 6016).

79/409/EEC. Council Directive 79/409/EEC of 2 April 1979 on the conservation of wild birds.

92/43/EEC. Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora.

97/266/EC: Commission Decision of 18 December 1996 concerning a site information format for proposed Natura 2000 sites.

DECISION No 2455/2001/EC OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 20 November 2001 establishing the list of priority substances in the field of water policy and amending Directive 2000/60/EC.

2008/32/EC. DIRECTIVE OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL 11 March 2008 amending Directive 2000/60/EC establishing a framework for Community action in the field of water policy, as regards the implementing powers conferred on the Commission.

7 ToR e) Evaluation of management schemes

e) Conduct a detailed, quantitative, evaluation of a limited number (2 or 3) of management schemes to assess the extent to which they actually reflect the high level definitions and are supported by a scientific evidence base. Possible case studies include the Baltic Sea Action Plan and the Norwegian Integrated Management Plan for the Lofoten-Barents Sea area. This would include the last four years of ecosystem-based management schemes. The plans should be assessed using a common framework.

7.1 General approach

7.1.1 What is the concern?

Large scale "ecosystem management plans" (or some variant of that name) are becoming more common (examples Barents Sea [[see Section 7.2], Baltic Sea [see Section 7.3], Puget Sound http://www.psp.wa.gov/aa_action_agenda.php; Eastern Scotian Shelf http://www.mar.dfo-mpo.gc.ca/oceans/e/essim/essim-intro-e.html; Chesapeake Bay http://www.chesapeakebay.net/]. In each case at an early stage in the process of producing the plan is some form of comprehensive review of science information available for the system. All cases follow the science review with an extended process of engagement with stakeholders, policy experts and other parties with roles in the development of the plans. Concern has arisen that by the end of the development process, links between the science foundations that were assembled at the start of the process and the provisions of the draft final plans may have weakened or been distorted.

The distortion could be of various forms. One type of possible distortion is "sins of omission": features considered by the science review important to conservation and sustainable use of the ecosystem(s) covered by the plan may not be well covered by objectives or provisions of the draft final plan. Another type of possible distortion of "sins of commission": objectives or provisions have been added to the draft final plans for reasons given as scientific, but the basis for those objectives or provisions is hard to find in the science information that was input to the start of the process. A third possible type of distortion is direct contradictions: objectives or provisions in the draft final plan are incompatible with information available in the science that input to the process.

The three types of distortions can be addressed in different ways. Omissions can be addressed by augmenting the draft final plan with additional objectives or provisions to fill the gaps. Commissions can either be removed or more likely given justifications on social, economic, or cultural grounds, such the proper reason(s) for the inclusion of the objective or provision is transparent. Direct contradictions should be fixed wherever possible, such that the objectives and provisions of the ecosystem plans are actually consistent with sound science. When there is a social or economic imperative that in the process of developing the ecosystem plan was considered to outweigh the ecosystem concerns, the plan has to at least call attention to the incompatibility between the objective or provision and the ecological information, and clarify the associated risks in a transparent manner.

7.1.2 The approach

The approach taken in this Term of Reference was:

1) For each candidate plan:
   - Extract the ecological economic and social objectives set out in the plans;
   - Extract the management measures and policy provisions intended to achieve the objectives;
   - Extract any other provisions which are ecological in nature and from the perspective of WGECO ought to have a basis in science.
2) Once these were extracted, we considered what type/level of science information should have been available, and if the objective or measure was to be justified by a science basis.
3) We review what types of science information could be pulled together for ground-truthing against the science that, based on the conclusions in 2) should have been available, as well as an inventory of the science that the plans themselves say was the basis for their provision.
4) Intersessionally, efforts will be made to pull together the data identified in 3, to have available for WGECO in 2011 to continue work on this topic (Section 7.4).

7.2 Barents Sea

7.2.1 General introduction to the Barent Sea Ecosystem Plan

7.2.1.1 Short history of the development of the plan

In 2006, the Norwegian government launched a White Paper for an ecosystem approach management plan for the Norwegian part of the Barents Sea, including the fishery protection zone around Svalbard and the sea areas off the Lofoten Islands (Anon., 2006) (Figure 7.2.1.1.1). Following international guidelines for ecosystem-based management, the plan provides an overall framework for managing all human activities (oil and gas industry, fishing, and shipping) in the area to ensure the continued health, production, and function of the Barents Sea ecosystem (Olsen *et al.*, 2007). The plan follows the main principles of the FAO guidelines for ecosystem approach -based fisheries management (Garcia *et al.*, 2003; FAO, 2005) and the implementation rules laid down recently by the UN (Ridgeway and Maquieira, 2006). The overall aim of the plan is to safeguard the marine ecosystem to ensure long-term value to mankind.

The plan is based on an assessment of the current and anticipated impact of human activities and of the interactions between them, taking into account deficits in current knowledge of ecosystem state and dynamics. To monitor the overall development of the Barents Sea's state of health, a set of indicators with reference levels was established.

Later in the same year, the Norwegian parliament passed a comprehensive integrated ecosystem approach-based management plan for the Barents Sea and the sea areas off the Lofoten Islands (Anon., 2006), covering all areas offshore of 1 nautical mile of the coast within the Norwegian EEZ, as well as the fishery protection zone around the Svalbard archipelago (Figure 7.2.1.1.1).

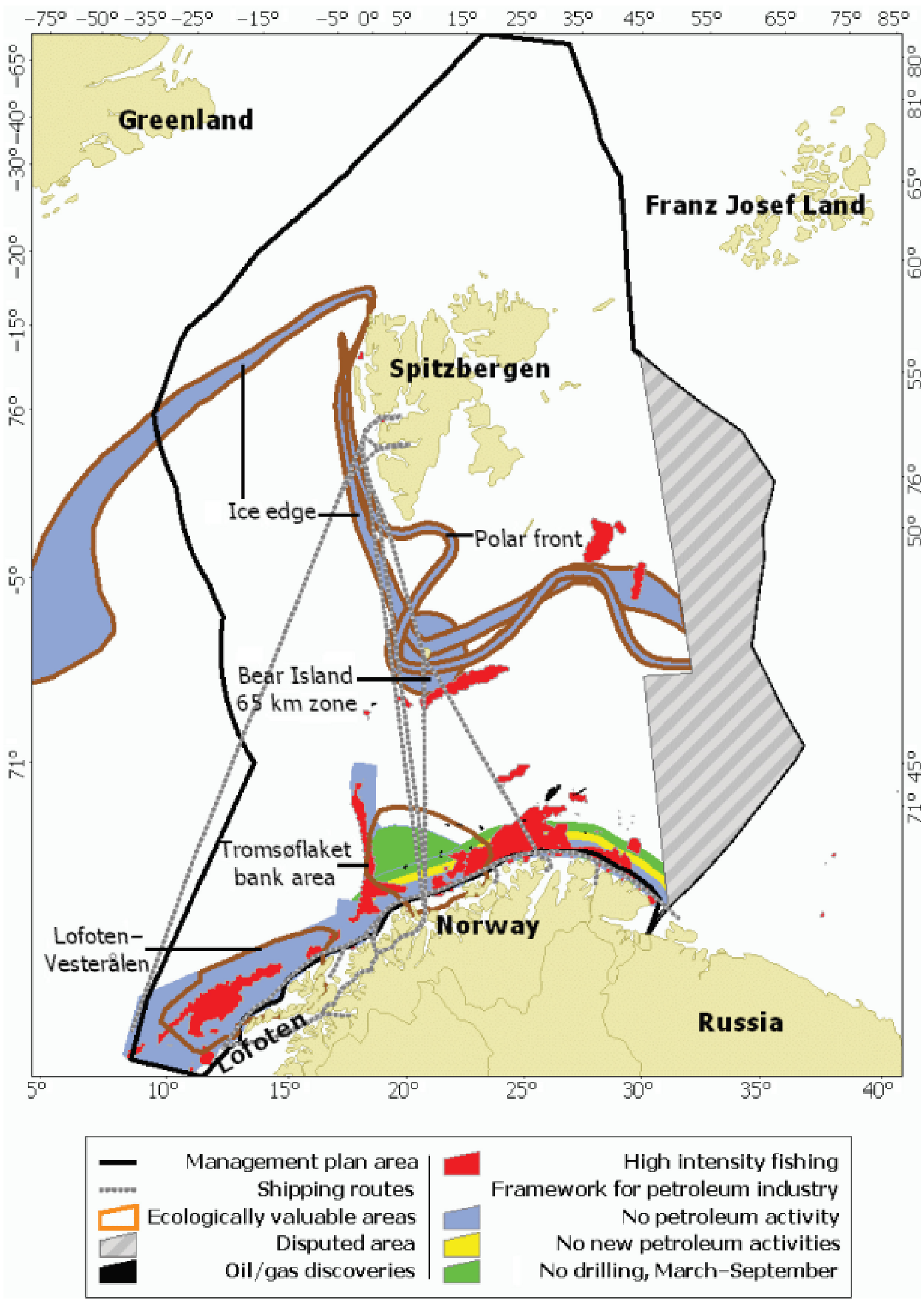


Figure 7.2.1.1.1. Area covered by the ecosystem-based management plan for the Barents Sea, showing the main fishing areas, shipping lanes, and the area-based framework for hydrocarbon extraction (2006–2010), together with the particularly valuable and vulnerable areas (from Olsen *et al.*, 2007).

7.2.1.2 The science input to the plan

Status reports have been prepared since 2002 by governmental management and research institutions, covering the state of the marine environment, especially valuable areas, to some extent the coastal zone, and for human activities: fisheries, aquaculture, and shipping. The initial reports uncovered major gaps in current knowledge. This was Step 1 in the plan-process. Out of nine basic scientific reports, four were related to ecology and natural resources, while the other five focused on fisheries, aquaculture, maritime transportation and human society.

These baseline studies were followed by sector-related study programmes and reports on External pressures (four reports), Fisheries (three reports) and Risk assessment, mainly related to the petroleum sector and acute oil spills (three reports).

The sectorial reports were then the basis for four cross-sectorial reports: i) Proposed management targets for the Barents Sea–Lofoten area; ii) Vulnerable areas and conflicts of interest; iii) Impacts of overall pressures on the Barents Sea–Lofoten area; iv) the knowledge deficiencies identified for the Barents Sea.

Parallel with the sectorial reports, a set of potential indicators were developed (von Quillfeldt and Dommasnes, 2005), based on high-level management goals. These covered climate, ice edge, and the functional levels of the ecosystem: phytoplankton, zooplankton, commercial fish species, non-commercial fish species, benthic organisms, marine mammals, seabirds, alien species, threatened and vulnerable species, and hazardous elements. Possible operational objectives were also clarified. No GES criteria were defined.

7.2.1.3 The consultation process that generated the plan

The work to prepare the plan was led by a government-appointed steering group chaired by the Ministry of the Environment, with representatives from other relevant ministries. To achieve transparency, all reports and other documents were made available through the internet, and stakeholders were invited to comment at several steps in the process.

Development followed a three-step process (Figure 7.2.1.3.1), not unlike the Eastern Scotian Shelf Integrated Management project (O'Boyle *et al.*, 2005). In Step 1, the status reports (the basic science reports) were prepared. Since the initial reports uncovered major gaps in current knowledge, a key principle was to use caution in the face of uncertainty. Also, the plan had to be dynamic to allow the evaluation of new knowledge as it became available.

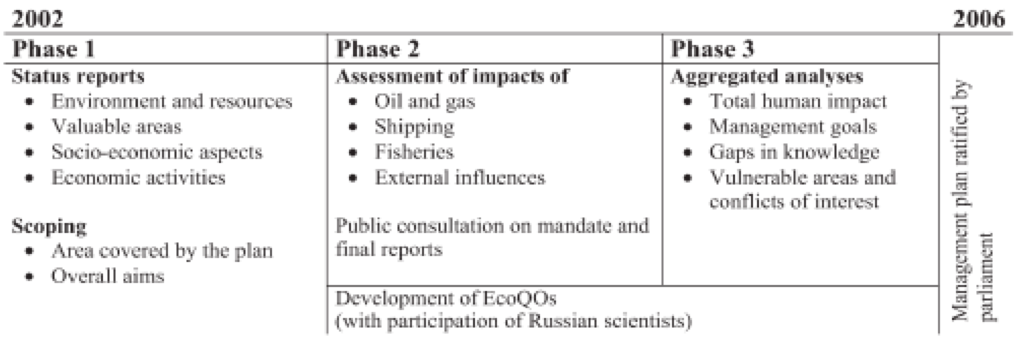


Figure 7.2.1.3.1. Three-phase development of the management plan for the Barents Sea, 2002–2006. The work was led by a steering group with representatives of four ministries, and the analyses and assessments were carried out by government directorates and research institutes (from Anon., 2006).

Step 2 represented an analytical phase based on Step 1, the sectorial reports. Four extensive government-funded Environmental Impact Assessments (EIAs) were carried out, covering the impact of fisheries, shipping, hydrocarbon extraction, and external pressures (e.g., pollution) on the environment, resources, and local communities. To ensure consistency and compatibility among the EIAs, a set of common variables was used to compare impacts among sectors, largely an *ad hoc* approach comparable with the hierarchical process used in Canada (O'Boyle and Jamieson, 2006). Impacts were assessed in relation to the starting situation (i.e., 2003) and in relation to expected future impacts up to 2020, with uncertainty obviously increasing over time.

In Step 3, the EIA results were brought together and analysed in more detail, focusing on: (i) the total impact of all human activities combined, both for the current situation and up to 2020; (ii) area conflicts among human activities, and between human use and ecologically valuable areas; (iii) the definition of high-level management goals required for implementation; and (iv) identification of gaps in current knowledge. This gave the four cross-sectorial reports.

Based on the reports, an intergovernmental group finally put together a White Paper, which was circulated a draft for hearing among scientific institutions, stake-holder-sectorial policy-makers and relevant organisations. Finally, with the comments from the hearing, the final management plan was presented by the government in 2005–2006 and approved by the parliament in 2006.

The management plan has led to changes in the Norwegian sampling and survey-programmes, the process of reporting monitoring of the state of the ecosystem, as well as a setting of the data reports.

The plan will be evaluated and revised every four years. The first evaluation report is due April 15, 2010.

7.2.2 Inventory of the objectives

The Barents Sea management plan represents a synergy of former separate management regimes of fisheries, shipping and the hydrocarbon industries. It does not contain any details on specific management measures for human activities. The complete list of measures used in the plan, are given in Section 7.2.5.1.

Since the overall aim of the plan is to safeguard the marine ecosystem to ensure long-term value to mankind, a selection of high order objectives is particularly listed in the plan (Anon., 2006, English version):

- "*The purpose of this management plan is to provide a framework for the sustainable use of natural resources and goods derived from the Barents Sea and the sea areas off the Lofoten Islands (subsequently referred to as the Barents Sea–Lofoten area) and at the same time maintain the structure, functioning and productivity of the ecosystems of the area*".
- "*The Government considers it very important to safeguard the basic structure and functioning of the ecosystems of this area in the long term, so that they continue to be clean, rich and productive*".
- "*The plan is also intended to be instrumental in ensuring that business interests, local, regional and central authorities, environ-mental organizations and other interest groups all have a common understanding of the goals for management of the Barents Sea–Lofoten area*".

Olsen *et al*. (2007) summarized that the plan aims at the sustainable use of the ecosystem, within acceptable levels of pollution with reduced risk of accidental spills, with sufficient capacity and readiness to deal with accidents, and seafood that is safe for consumption, while safeguarding biodiversity.

Stiansen *et al*. (2009) has also summarized the main objectives and stated that one of the goals of the Barents Sea management plan has been:

- *"Management of the Barents Sea–Lofoten area will ensure that diversity at ecosystem, habitat, species and genetic levels, and the productivity of ecosystems, are maintained. Human activity in the area will not damage the structure, functioning, productivity or dynamics of ecosystems."*
- *"Releases and inputs of pollutants to the Barents Sea–Lofoten area will not result in injury to health or damage the productivity of the natural environment and its capacity for self-renewal. Activities in the area will not result in higher levels of pollutants."*

By the time the management plan was implemented, the steering group had cut back on the list of suggested indicators, decided on reference levels in accordance with the suggestions from the scientists and approved of the threshold for actions (Annex 7.2.5.2.). The rejected indicators in 2005–2006 and indicators suggested during the evaluation process of 2009–2010 are in this Annex. The plan was provided with detailed objectives to be met (Table 7.2.2.1). Both well-established data-series and undeveloped indicators were included, to provide a representative selection for the ecosystem functions and the natural pressures such as climate, water quality and ice cover. A selected set is presented in Table 7.2.2.2.

Table 7.2.2.1. A sector disaggregated list of objectives given in the Barents Sea management plan (Anon, 2006). The text blocks are lifted directly from the text of the management plan. Additionally, subjective interpretations for driving motivation for the objectives are suggested: Yes (Y= relevant and beneficial), No (N= no relevance or possibly costly) and Partially (P= indirectly relevant).

SECTOR-RELATED	OBJECTIVES	MOTIVATED BY NATURE CONSERVAT-ION	MOTIVATED BY SOCIO-ECONOMY
Conservation of marine habitats	A representative network of protected marine areas will be established in Norwegian waters, at the latest by 2012. This will include the southern parts of the Barents Sea–Lofoten area.	Y	P
Biodiversity	Management of the Barents Sea–Lofoten area will ensure that diversity at ecosystem, habitat, species and genetic levels, and the productivity of ecosystems, are maintained. Human activity in the area will not damage the structure, functioning, productivity or dynamics of ecosystems.	Y	Y
Valuable and vulnerable areas	Activities in particularly valuable and vulnerable areas will be conducted in such a way that the ecological functioning and biodiversity of such areas are not threatened.	Y	N
Habitats	In marine habitats that are particularly important for the structure, functioning, productivity and dynamics of ecosystems, activities will be conducted in such a way that all ecological functions are maintained.	Y	N
Species management, inc. harvesting	Naturally occurring species will exist in viable populations and genetic diversity will be maintained.	Y	N
	Harvested species will be managed within safe biological limits so that their spawning stocks have good reproductive capacity.	Y	Y

SECTOR-RELATED	OBJECTIVES	MOTIVATED BY NATURE CONSERVAT-ION	MOTIVATED BY SOCIO-ECONOMY
	Species that are essential to the structure, functioning, productivity and dynamics of ecosystems will be managed in such a way that they are able to maintain their role as key species in the ecosystem concerned.	Y	Y
	Populations of endangered and vulnerable species and species for which Norway has a special responsibility will be maintained or restored to viable levels. Unintentional negative pressures on such species as a result of activity in the Barents Sea–Lofoten area will be reduced as much as possible by 2010.	Y	N
	The introduction of alien species through human activity will be avoided.	Y	P
Maritime transport	The risk of acute oil pollution from maritime transport in the area of the management plan is at present lower than for other Norwegian sea areas. One of the main aims of Norwegian risk management through maritime safety and oil spill response measures is to keep the risk of damage to the marine environment and living marine resources at a low level.	Y	Y
Petroleum activities	One of the main goals of Norway's risk management of petroleum operations is to reduce the environmental risks to the minimum practical level.	Y	Y
Pollution	Releases and inputs of pollutants to the Barents Sea–Lofoten area will not result in injury to health or damage the productivity of the natural environment and its capacity for self-renewal. Activities in the area will not result in higher levels of pollutants.	Y	Y
	The environmental concentrations of hazardous and radioactive substances will not exceed the background levels for naturally occurring substances and will be close to zero for man-made synthetic substances. Releases and inputs of hazardous or radioactive substances from activity in the area will not cause these levels to be exceeded.	Y	Y
	Operational discharges from activities in the area will not result in damage to the environment or elevated background levels of oil or other environmentally hazardous substances over the long term.	Y	Y
	Litter and other environmental damage caused by waste from activities in the Barents Sea–Lofoten area will be avoided.	Y	N
	Fish and other seafood will be safe and will be perceived as safe by consumers in the various markets.	N	Y

SECTOR-RELATED	OBJECTIVES	MOTIVATED BY NATURE CONSERVAT-ION	MOTIVATED BY SOCIO-ECONOMY
	The risk of damage to the environment and living marine resources from acute pollution will be kept at a low level and continuous ef forts will be made to reduce it further. Activity that involves a risk of acute pollution will be managed with this objective in mind.	Y	Y
	Maritime safety measures and the oil spill response system will be designed and dimensioned to effectively keep the risk of damage to the environment and living marine resources at a low level.	Y	Y

Table 7.2.2.2. Examples of specific objectives of the management plan lifted from Olsen *et al.* (2007). State of enforcement is based on the state-of-art 2010.

SECTOR-RELATED	SOME LELECTEDSPECIFIC OBJECTIVES	STATE OF ENFORCEMENT
Ecosystem conservation	Prevention of the introduction of alien species	No actions
	Protection of valuable and threatened habitats	Enforced
Harvesting	Implementation of ecological measures in fishery management based on an increased use of multispecies assessment tools, and aimed at a reduced bycatch of fish, seabirds, and marine mammals, and fewer effects on bottom fauna	in development, not implemented in the management tools
	Increase in the number of target species managed sustainably and under a precautionary approach	Not yet in place, intended for fish stock in recovery
	Measures against illegal, unregulated, and unreported (IUU) fishing	enforced successfully
	Global ban on selling IUU fish	International initiatives taken
	Closer cooperation with the EU, Russia, and others nations	enforced
	Surveillance, and including the prosecution of fishers violating,	enforced
	Existing rules (e.g. discarding, catching undersized fish, unacceptable modifications to gear);	enforced
Petroleum activity	Hydrocarbon industry to operate under a zero emission policy;	enforced, and claimed to be abided by the oil companies
Maritime transportation	Shipping lanes outside territorial waters to reduce the risk of collision and to allow increased time for remedial action	enforced
Pollution	Further preventative measures against pollution, both locally and regionally	in development

7.2.3 Organization of the plan

The organization of the management plan is shown in Figure 7.2.3.1. There are quite a few different groups involved in the running of the plan. Some of the key groupings are:

- The advisory group on monitoring which assists in the coordination of the system proposed by the Government for monitoring the state of the environment.
- The forum on environmental risk management focusing on acute pollution in the area, which will provide valuable input to environmental risk assessments.
- The management forum responsible for the coordination and overall implementation of the scientific aspects of ecosystem-based management of the Barents Sea-Lofoten area.
- The reference group for the work on the ecosystem-based management regime that represents the various interests involved, including business and industry, environmental organisations and Sami interest groups.

Scientific gaps are dealt with in an operational manner through an annual gap analysis (Figure 7.2.3.1). The scientific gaps recognized are passed on from the steering group to the funding agencies and used during the prioritization in funding of new research projects. Also the activities will be recognized as important for the scientific institutions that take part in the Steering Group. This group is lead by the Ministry of the Environment.

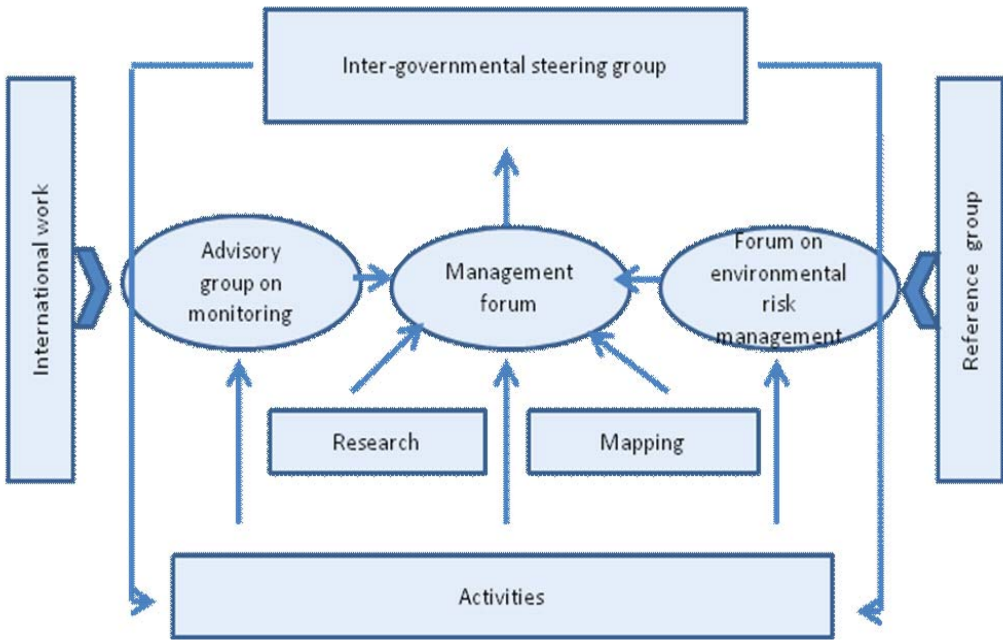


Figure 7.2.3.1. Overview of the elements of the system for implementing the management plan.

While the management legislation is still located sector-wise, the most pronounced change in the management of the region is connected to the establishment of the cross-sectorial monitoring and advice groups and the ecosystem-related list of indicators. The monitoring of the ecosystem includes:

- The set of indicators with reference levels and action thresholds are set in place to manage the system based on the high level management objectives (Section 7.2.5.2).

Experience by 2010: Some of the under-developed indicators are by 2010 more developed while others are still to be made operational for management use. No changes have been made in the list since 2006, but new suggestions are reported as part of the

2010 evaluation reports to fill gaps of information needed to enhance the insights in the ecosystem processes.

- Monitoring of the natural elements of the ecosystem and pollution is organized through an annual report.

Experience by 2010: The report mainly focuses on presenting the status of the indicators in relation to the reference levels that are given in the plan. For most of the indicators threshold values for when an action should be taken have been developed. However, the monitoring group also strives to evaluate the functioning and structure of the ecosystem, by releasing examples and descriptions on how selected indicators interact. The annexed indicator list presents known interaction-links. A methodology to monitor and report on the interactions between the indicators is not yet implemented.

- Monitoring of the human impact levels related to fisheries, shipping and hydrocarbon activities has been requested by the monitoring group in relation to the revision of the plan in 2010.

Experience by 2010: Such systematic monitoring and reports into the management plan are not fully developed. It is partly met by the work of the Forum for Environmental Risk Management, but is not included in the monitoring of the state of the ecosystem.

7.2.4 Inventory of the provisions in the plans for management measures, policies, etc.

Measures for protection are essentially temporary tools to prevent negative consequences of human actions on an area, ecosystem component, or species when threats are severe, but do not necessarily provide permanent refuge (Section 7.2.5.1, taken from Anon, 2006). A central concept of the plan is that it is based on science and takes a precautionary approach, implying a need for revision as new knowledge becomes available. The plan represents a synergy of previously separate management regimes: management of fisheries, shipping, and the hydrocarbon industry are brought together under one umbrella to coordinate efforts and to achieve a healthy ecosystem. Still, these management regimes are to continue to function and act within their own fields.

In practice, achieving measurable improvements in all these sectors is the main challenge, and these are envisaged by implementing: (i) area-based management to resolve conflicts between activities and protecting the environment; (ii) continuation of established management measures regulating the various activities; (iii) implementation of EcoQOs (Section 7.2.5.1); and (iv) increased focus on international cooperation, regionally and globally. International cooperation has been followed up by the release of the Norwegian-Russian Environmental Report on the Barents Sea, 2009.

7.2.5 Technical annexes for the Barents Sea Ecosystem Plan

7.2.5.1 List of measures from the Anon, 2006

Topic-related	Measures according to the Barents Sea management Plan
Ecosystem-based harvesting of living marine resources	Continue the development of an ecosystem-based management regime, in order to ensure an integrated approach to the management of the commercial species and an assessment of how this affects the ecosystem as a whole. This also requires taking into account vulnerable and endangered species and their nutritional needs.

Topic-related	Measures according to the Barents Sea management Plan
	Increase the proportion of commercially exploited stocks that are surveyed, monitored and harvested in accordance with existing management strategies, including management targets.
	Set precautionary reference points for all the spawning stocks that are exploited commercially, particularly stocks that are being rebuilt to sustainable levels.
	Reinforce control measures to ensure that harvesting takes place in accordance with the TACs.
Illegal, unreported and unregulated fishing (IUU)	Work towards arrangements that will make it impossible for fish caught during IUU fishing to be sold or landed in any part of the world,
	Cooperate more closely with fisheries authorities in other countries, particularly Russia and the EU
	Seek to conclude agreements on fisheries control with countries with which no such agreements exist, effectively follow up and investigate cases of IUU fishing
	Strengthen overall efforts in this field (grants to the Directorate of Fisheries and the Norwegian Coast Guard have been increased for this purpose),
	Set up a special task force in the Directorate of Fisheries with the responsibility for uncovering economic crime.
Unintentional pressures on benthic fauna	Continue systematic surveys of the seabed under the MAREANO programme with a view to full implementation of this programme in the Barents Sea–Lofoten area by 2010,
	Through ecosystem surveys, initiate systematic monitoring of the benthic fauna in the Barents Sea,
	Survey the Tromsøflaket bank area in order to identify sponge communities,
	Compare the sponge communities on Tromsøflaket with similar communities elsewhere with a view to possible protection,
	Ensure satisfactory protection of coral reefs in the Barents Sea–Lofoten area, for example by establishing a cross-sectoral national action plan for coral reefs,
	Continue the work of surveying coral reefs and providing adequate safeguards for new reefs that are discovered, and regularly provide the fishing fleet and other operators with updated information on new coral reefs,
	Increase its focus on building up better and more complete statistics for shipping by systematic compilation of information from existing databases (such as AIS data, the pilot database, satellite tracking data for fishing vessels, data from other satellite-based systems, Safe Sea Net data, etc.). One purpose is to improve the input data for risk analyses with a view to preventing and detecting acute spills and making it possible to identify the sources.
	Further develop gear that is towed along the seabed in order to reduce bycatches and destruction of the benthic fauna.
Unintentional bycatch of seabirds	Contribute to long-term build-up of the knowledge base on seabird populations through the SEAPOP seabird monitoring programme. This will give the various sectors which affect the marine environment, including seabirds, a better basis for implementing any necessary measures.
	Assess the need for regulatory measures in the fisheries in line with up-to-date information on the distribution of seabirds (where and when) and on their need for protection.
	Make suitable arrangements to obtain better documentation of the bycatch problem.

TOPIC-RELATED	MEASURES ACCORDING TO THE BARENTS SEA MANAGEMENT PLAN
Introduction on alien species	Play a part in ensuring that the international rules on the introduction of alien species are complied with and strengthened.
	Take steps to improve knowledge of alien species and the risks associated with their introduction through a cross-sectoral national strategy for alien species that is currently under preparation with a view to completion in the course of 2006.
	Ratify the Ballast Water Convention and provide the necessary legal basis for taking measures to implement it.
Endangered and vulnerable species and habitats	Draw up and implement action plans for selected habitats, groups of species and species in the Barents Sea–Lofoten area in the period up to 2010 as part of its efforts to halt the loss of biodiversity by 2010 (see Report No. 21 (2004–2005) to the Storting).
	Contribute to the development of a regional ballast water strategy for the OSPAR area in cooperation with HELCOM.
	In connection with a separate white paper on management of the red king crab, as set out in the budget proposal for 2006 from the Ministry of Fisheries and Coastal Affairs, consider whether a limit should be set north of which unrestricted fishing for red king crab may be introduced.
	Implement national measures to fulfil the provisions of the Convention: this will include an assessment of whether it is necessary to establish special zones for ballast water exchange, taking into consideration transport routes and risks, and the establishment of monitoring and notification routines and emergency response plans where there is a danger of acute exposure.
Acute oil spill, petroleum, and maritime transportation	Increase its focus on building up better and more complete statistics for shipping by systematic compilation of information from existing databases (such as AIS data, the pilot database, satellite tracking data for fishing vessels, data from other satellite-based systems, Safe Sea Net data, etc.). One purpose is to improve the input data for risk analyses with a view to preventing and detecting acute spills and making it possible to identify the sources.
	Continue its work on maritime safety and oil spill response measures as set out in a recent white paper on maritime safety and the oil spill response system (Report No. 14 (2004–2005) to the Storting).
	Increase its focus on building up better and more complete statistics for shipping by systematic compilation of information from existing databases (such as AIS data, the pilot database, satellite tracking data for fishing vessels, data from other satellite-based systems, Safe Sea Net data, etc.). One purpose is to improve the input data for risk analyses with a view to preventing and detecting acute spills and making it possible to identify the sources.
	Cooperate with Russia on the analysis and identification of the types of oil transported by ship along the coast in the area covered by the management plan and evaluate the need to establish a data bank for all these types of oil.
	Introduce traffic restrictions in the protected area on Svalbard for ships with heavy bunker oil on board.
	By transferring more responsibility to the business sector within the existing frameworks and legislation, ensure that training modules adapted to the specific environmental and operational conditions in the Barents Sea–Lofoten area are developed.
	Strengthen the meteorological observation base.
	No petroleum activities will be initiated within a 65 km zone round Bjørnøya.
	The Bjørnøya nature reserve will be expanded to the 12-nautical-mile territorial limit.

Topic-related	Measures according to the Barents Sea management Plan
	No petroleum activities will be initiated in or near the marginal ice zone and the polar front.
	No petroleum activities will be initiated within a zone stretching 35 km outwards from the baseline from the Troms II petroleum province along the coast to the Russian border.
	No new petroleum activities will be initiated in the zone 35–50 km from the baseline, with following exceptions: petroleum activities in areas for which production licences were awarded in the 19th and earlier licensing rounds may be continued; new announcements and licence awards are permitted in predefined areas in mature parts of the shelf (APA area), and there will be openings for development of additional resources in these areas. The question of petroleum activities in the 35–50 km zone will be considered when the management plan is revised in 2010.
	No exploration drilling will be permitted in oil-bearing formations in the zone 50–65 km from the baseline nor at Tromsøflaket outside 65 km from the baseline, in the period 1 March–31 August.
	No petroleum activities will be initiated in Nordland VII and Troms II during the current parliamentary period. The question of petroleum activities in these areas will be considered when the management plan is revised in 2010.
	The SEAPOP programme (Seabird Population Management and Petroleum Operations) will give priority to surveys in the Lofoten and Vesterålen Islands and Eggakanten area (stretching northwards from the Tromsøflaket bank area).
	The MAREANO programme to develop a marine areal database for Norwegian waters will give priority to surveys in the Lofoten and Vesterålen Islands and Eggakanten area (stretching northwards from the Tromsøflaket bank area)..
	Geological surveys will be carried out in the area under the direction of the Petroleum Directorate. This will include acquisition of seismic data.
	No petroleum activities will be initiated in Nordland VI, Eggakanten area during the current parliamentary period (2006–2010).
	In areas of the southern Barents Sea where no special requirements or restrictions apply in accordance with the points above, no licence-specific conditions will apply apart from the requirement for zero discharges to the sea under normal operating conditions.
	This means that licence-specific conditions previously laid down, for example on exploration
Long-range pollution	Give priority to the work of following up the Strategic Approach to International Chemicals Management (SAICM), which has been adopted by the United Nations Environment Programme (UNEP).
	Work towards the elimination of mercury releases as far as possible through a binding global convention.
	Propose the inclusion of new hazardous substances in international agreements such as the Stockholm Convention on Persistent Organic Pollutants (POPs), as appropriate.
	Ensure that efforts to reduce the use and discharge of hazardous substances are given high priority in development cooperation and in cooperation with Russia.
	Seek to play an active part in efforts to ensure that the proposed new EU regulatory framework for the Registration, Evaluation and Authorisation of Chemicals (REACH) affords the best possible protection of the environment, consumers and employees.

Topic-related	Measures according to the Barents Sea management Plan
	Propose more persistent organic pollutants for inclusion in the Aarhus Protocol on Persistent Organic Pollutants.
	Participate actively in the revision of the Gothenburg Protocol to Abate Acidification, Eutrophication and Ground-level Ozone.
	Ensure that marine organisms are taken into consideration in processes related to the development of risk management tools in the OSPAR Commission, the EU and other international fora.
Pollution in general	Consider establishing an environmental specimen bank for the Barents Sea–Lofoten area to make it possible to re-analyse specimens as better methods of detecting hazardous substances are developed and new substances are found in the environment, and to determine reference values/background levels for new hazardous substances and establish time trends.
	By holding the industries accountable for the pollution they generate, ensure the development of working methods to further reduce the use and discharge of hazardous substances and the development of less hazardous substances with equally good operational performance.
	Strengthen control measures and legal follow-up in cases of illegal discharges/pollution from installations and vessels in the area.
	Take the initiative within the framework of the International Maritime Organization (IMO) for the development of better methods, including the development of emission factors, for estimating operational discharges from ships.
	Make the Seaworthiness Act applicable to Svalbard and pursuant to the Act, make the regulations on the prevention of pollution from ships applicable to foreign ships in the waters around Svalbard.
	Consider whether there are grounds for requesting the IMO to accord the Barents Sea the status of Special Area (SA) under Annex I and Annex V of the Marpol Convention 1973/1978, in order to be able to invoke the more stringent rules of the IMO regarding discharges of oil and garbage from ships which apply in Special Areas.
	Take the initiative vis-à-vis the IMO on a revision of Annex V on prevention of pollution by garbage from ships with a view to implementing rules with which compliance is easier to control and which take into account the new technologies in the field of waste management.

7.2.5.2 List of indicators

Indicators to be monitored according to the management plan	Data type	Reference level	Treshold, when action is to be considered	Reated directly to indicators (based on subjective discussions without checking for references)
1. Ice cover	Distribution (twice a year)	Average 1997–present	None described	2, 3, 4, 8, 10
2. Water quality	Temp., salinity, nitrate, silicate	Average 1997–present	None described	1, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12
3. Transportion of Atlantic water into the Barent Sea	Volume	Average 1997–present	None described	1, 2, 4, 5, 6, 7, 8, 9, 10, 11, 12
4a. Timing of the spring bloom	Observation data,	None	None described	1, 2, 3, 4
4b. Phytoplankton	Biomass and Production	Average the last 10 years	None described	1, 2, 3, 4, 5
4c. Chlorophyl a	Measured data, related to season and ice cover	Historical data	None described	1, 2, 3, 4
5a. Zooplankton	Biomass calculations: weight, length, fatty acid profile	Average the last 10 years	None described	1, 3, 4, 6, 7, 10
5b. Zooplankton	Species counts	Historic levels	None described	1, 3, 4, 6, 7, 10
6a. and b. Young herring and Blue whiting	Biomass, Distribution	Historic levels	None described	2, 3, 5, 7
7a and b Spawning stocks	Total stocks and distribution of NEA cod, Capelin	Mpa	Below Mpa	2, 3, 5, 6, 9, 10
8a, b and c. Fish stocks under rebuildinng	Assessments, Greenland halibut, Red fish, Deepwater red fish	Mpa (not yet established)	Below Mpa	2, 3, 9, 12
9a. Benthos	Species, Individual density, Biomass	Under development	Under development, Relating new data to historic data	1, 2, 3, 4, 5, 7, 8
9b. Benthos sesille species	Observation data,	Distribution data and state of described locations	High degree of physical damage	1, 2, 3, 4, 5, 7, 8
9c. Red king crab	Estimated density, Distribution, Total population	Known distribution	Dispersal into new areas	2, 9, 11
10a, b, and c. Seabird, Population development	Observation data,, common guillemot, thick-billed guillemot, Atlantic puffin	Average last 10 years + historic data	Negative development related to average last 10 year and historic data	2, 3, 5, 6, 7, 12

Indicators to be monitored according to the management plan	Data type	Reference level	Treshold, when action is to be considered	Reated directly to indicators (based on subjective discussions without checking for references)
Incl. Seabird Hatching success	Observation data, common guillemot, thick-billed guillemot, Atlantic puffin	Sufficient success to sustain the species + historical data	Failures last five years	2, 3, 5, 6, 7, 12
Incl. Seabird; Adult survival rate	Observation data, common guillemot, thick-billed guillemot, Atlantic puffin	Sufficient success to sustain the species + historical data	20% reduction in population within five years	2, 3, 5, 6, 7, 12
Incl. Seabird, Food availability	Observation data, common guillemot, thick-billed guillemot, Atlantic puffin	Not described, linked to zoo-plankton data-registration	None described	5, 6, 7
10d. Spacial distribution of sea mammals	Scientific survey-data	Average last 10 years + historic data	None described	1, 2, 3, 5, 6, 7
10e. Bycatch harbour porpoises	Selected fishermen diaries	Data published by Bjørge *et al* 2006a; b; c	When bycatch increase above reference value (average bycatch in 2006–2008)	
11a. New alien species	No monitoring	Historic data	When discovered in the region	1, 2, 3
11b and c..Observed alien species	Red king crab and Snow crab	Historic data	When discovered in new areas of the region	2, 3, 9
12. Red-listed species	Population estimate	Historic data	Below conception of populations below reproductive levels	1, 2, 3. 8, 9, 10
13a. Littering	Weight of litter at the Soitsbergen coast	No litter	Litter present	8, 10
13b, c, d, f, g and h Toxicology, sea mammals, sea birds, fish, coastal cod, benthos, sediments	Heavy metal, environmental toxic in polar bear, walrus, bearded seals, hooded seals, harp seals, beluga whale	Natural levels	Steady increase over years, or sudden incrase from one sampling to next	7, 8, 9, 10
13i. Environmental pollutants in the upper sediment layer	JAMP, Environmental toxics	Natural levels	Steady increase over years, or sudden incrase from one sampling to next	7, 8, 9, 10

Indicators to be monitored according to the management plan	Data type	Reference level	Treshold, when action is to be considered	Reated directly to indicators (based on subjective discussions without checking for references)
13j and k.. Nuclear radiation levels in seaweed and sediments	Measurements of 137Cs, 99Tc, 239+240Pu (only seaweed)	Natural levels	Steady increase over years, or sudden incrase from one sampling to next	7, 8, 9, 10
13 l and m. Airborn and riverborn pollutants	Heavy metal, environmental toxic, pesticides, climatic gasses, spore elements	Analyses	Steady increase over years, or sudden incrase from one sampling to next	7, 8, 9, 10
Indicators suggested for monitoring but not included in the management plan				
Phytoplankton, nitrate/silikate	Analyses			
Fish mortality due to fishing	Calculations			
0-group indeces	Assessments			
NEA cod, stomach content	Specifications, species count, biomas			
Occurences of non-commercial fish species	Observations			
Ice bear populations development	Observations			
Sea mammals population development	Observations, walrus, harp seal, minke whale,			
Sea mammals, distribution	Occurence and count, whalerus, minke whale			
Migration of Atlantic benthic species into the Barents Sea	Observations			
Coastal benthic species at Spitsbergen	Observations			
Vulnerable or thratened species, of special concern and responsibility				
Effect on Iceland scallp societies by red king crab	Observations, sampling			

Indicators to be monitored according to the management plan	Data type	Reference level	Treshold, when action is to be considered	Reated directly to indicators (based on subjective discussions without checking for references)
Effects of red king crab on soft-bottom benthic societies (models)	Density calculations, Bruke data fra haneskjellundersøkelsene			
Introduced species by ships	Observations			
Toxicology of Iceland scallop, blunt gaper	Tissue sampling			
Indicators on pressures, suggested by the monitoring group for inclusion in the revised management plan Sunnannå *et al*. (2010).				
Fisheries activities				
Shipping activities				
Oil and gas related activies				
Other indicators , suggested by the monitoring group for inclusion in the revised management plan Sunnannå *et al*. (2010).				
Ice-related species				
Species endemic for the Spitsbergen area				

7.3 HELCOM Baltic Sea Action Plan

7.3.1 General introduction to the HELCOM Baltic Sea Action Plan (modified from Backer (2008) and Backer *et al*. (in press))

HELCOM is the governing body of the "Convention on the Protection of the Marine Environment of the Baltic Sea Area (HELCOM, 1974)" - more usually known as the Helsinki Convention. The HELCOM Baltic Sea Action Plan (BSAP; HELCOM, 2007a) is a multilateral Ministerial Declaration (adopted on 15 November 2007) in which the HELCOM contracting parties, coastal country governments and the European Commission, commit to actions to achieve a number of agreed ecological objectives and, eventually, a Baltic Sea in “good environmental status” by 2021. The BSAP is explicitly based on the Ecosystem Approach and includes a number of initial targets pertaining to HELCOM’s four main themes, i.e., eutrophication, hazardous substances, biodiversity and nature conservation and maritime activities, as well as indicators to measure progress toward the commitments.

The whole HELCOM process toward fulfilling the implementation of the Ecosystem Approach can roughly be divided into four parts: 1) an initial preparatory phase defining aspirational objectives (2003–2006), 2) a subsequent quantitative phase defining operational targets based on the objectives, 3) drafting the dedicated plan of actions (i.e., the BSAP) (2005–2007) and 4) implementation of the actions (2008–).

The preparatory process, initiated in 2003, scoped through stakeholder consultation specific issues to be included in the Action Plan by developing an overall vision, strategic goals reflecting the four identified priority issues as well as regional Baltic Sea Ecological Objectives for the future Baltic Sea (Fig. 7.3.1.1).

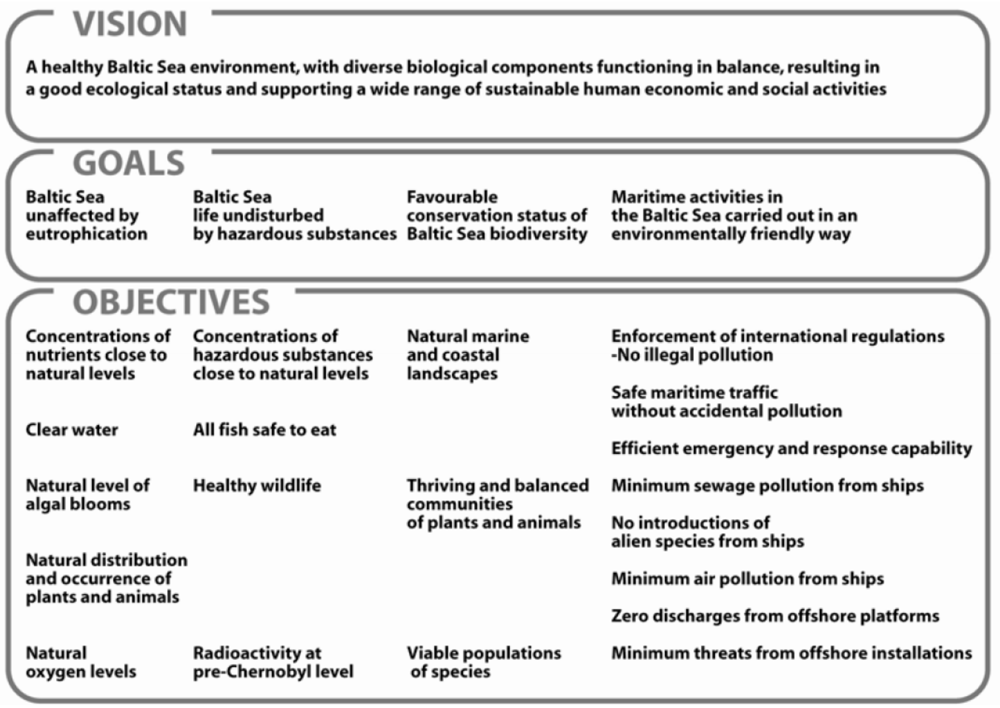


Figure 7.3.1.1. HELCOM overall vision and strategic goals reflecting the four identified priority issues as well as regional objectives for the future Baltic Sea (from HELCOM, 2007b).

Building upon the achieved political agreement on the Ecological Objectives, a number of operational targets, as well as management actions, were developed during 2005–2007. Since 2006 these activities focused around the development of a document that details how to implement the Ecosystem Approach in the Baltic Sea – the HELCOM Baltic Sea Action Plan (BSAP).

The concrete development of the four thematic segments of the Action Plan, representing the four Strategic Goals of HELCOM, was assigned to a number of lead countries/organisations including also NGOs. The final phases of the development comprised political discussions between Contracting Parties including coastal countries and the European Commission. In these discussions other stakeholders were participating as observers.

The Baltic Sea Action Plan

The BSAP includes the Baltic catchment area (see Figure 7.3.1.2) and distinguishes between measures that can be implemented at regional or national level, and measures that require implementation at EU or international levels. For the last two types of actions, the Action Plan commits the Contracting Parties to proactively reach regional consensus in the form of joint HELCOM inputs to relevant international processes. In the case of EU this includes the Common Fisheries Policy, Common Agricultural Policy and controls over the marketing and use of chemicals. Global measures include those for shipping taken within the International Maritime Organization, IMO. In addition to the preamble and the four thematic segments, the Action Plan includes chapters on assessment, financing and implementation/review. Further, ten technical recommendations including an amendment to the 1992 Helsinki Convention (parts of Annex III focusing on agriculture), initial environmental indicators and targets as well as a number of other documents were adopted as a part of the Action Plan.

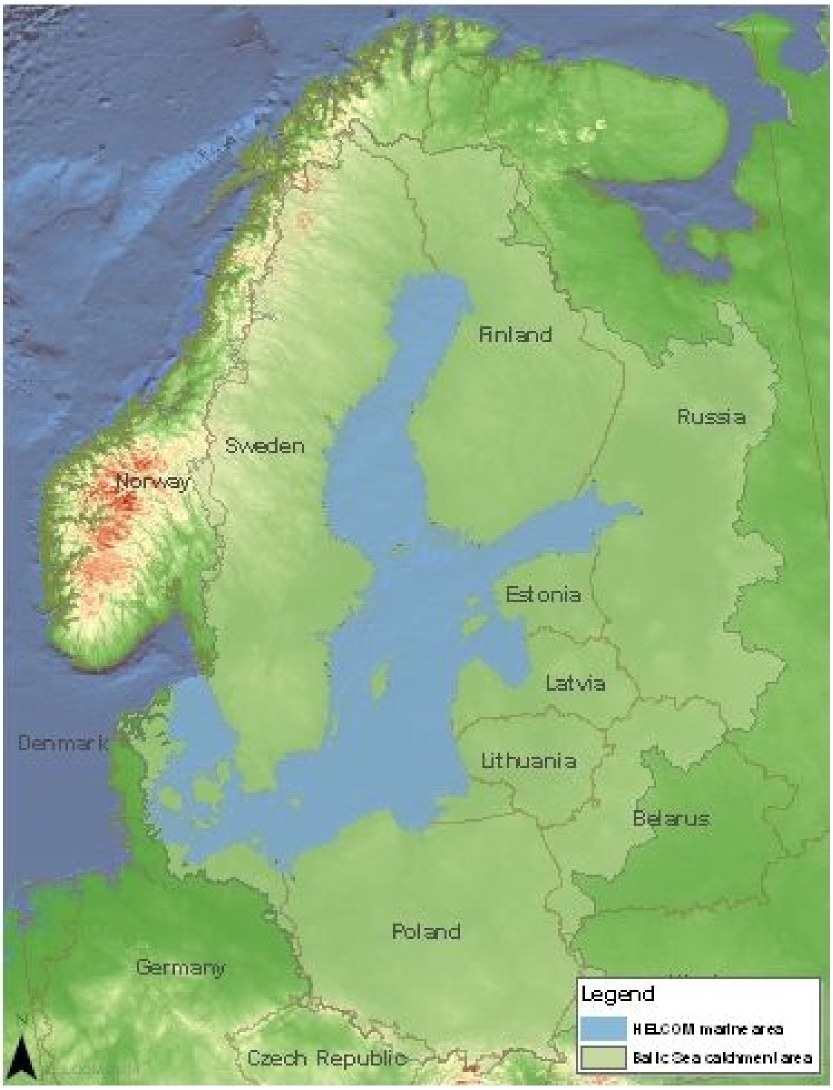


Figure 7.3.1.2. The HELCOM marine and catchment area (from: http://www.helcom.fi).

Although not quantitative as such, the adopted Ecological Objectives fulfilled an important practical and strategic function by defining, in the form of a political agreement, important characteristics requiring, and paving the way for, further quantitative definitions through indicators (Backer and Leppänen, 2008). Parallel HELCOM assessment activities (e.g., HELCOM, 2006) provided the necessary scientific consensus for quantitative targets. These were, and continue to be, developed in subsequent HELCOM assessment work (e.g., HELCOM, 2006; 2009a,b).

The BSAP is largely non-binding in the legal sense and therefore relies heavily on other fora to achieve its objectives. These fora include the MSFD, Natura 2000 directives, the Common Fisheries Policy, etc.

Over the coming years these will be measured and evaluated by regional HELCOM monitoring and assessment work. A HELCOM Ministerial Meetings to be held in

Moscow in 2010 and a subsequent meeting in 2013 are among the coming milestones where the eventual results will be revealed.

7.3.2 Inventory of the objectives

The HELCOM BSAP is divided into a preamble followed by separate sections covering eutrophication, hazardous substances, biodiversity and nature conservation and maritime activities. Included in the BSAP are also for example, sections on financing, implementation, tool development and awareness raising as well as a number of recommendations and technical annexes.

From the preamble it can be extrapolated that the BSAP focuses on overall objectives relating to the ecosystem approach and integrated management of human activities and on achieving a "Baltic Sea unaffected by eutrophication", "Baltic Sea with life undisturbed by hazardous substances", "Maritime activities carried out in an environmentally friendly way", all of which will lead to a "Favourable conservation status of Baltic Sea biodiversity".

In following thematic sections the BSAP describes the actions needed to be taken in order to achieve the above in relation to eutrophication, hazardous substances, biodiversity and nature conservation and maritime activities, respectively.

In order to limit the scope of this study and to include what has traditionally been the remit of WGECO the following analysis of explicit objectives, targets and indicators will be limited to the "Biodiversity and nature conservation segment of the HELCOM Baltic Sea Action Plan".

7.3.2.1 Inventory of the objectives pertaining to the biodiversity section of the BSAP (HELCOM, 2007a)

The specific strategic goal for the protection of biodiversity is to reach a "favourable conservation status of Baltic Sea biodiversity". This means that biodiversity is restored and maintained and all elements of the marine food-webs, to the extent that they are known, occur at normal abundance and diversity. The ecological objectives related to this goal are divided into marine and coastal landscape level, community level and species level, reflecting the Convention on Biological Diversity (CBD), in which the assessment is focused on variability "within species", "between species" and "of ecosystems".

In order to make the ecological objectives operational, concrete short-, middle- and long-term targets should be set and the progress toward these followed with indicators.

In order to reach favourable conservation status of *biodiversity*, HELCOM has adopted ecological objectives covering topics referring to:

- restoring and maintaining sea floor integrity at a level that safeguards the functions of the ecosystems;
- that habitats, including associated species, show a distribution, abundance and quality in line with prevailing physiographic, geographic and climatic conditions; and
- a water quality that enables the integrity, structure and functioning of the ecosystem to be maintained or recovered.

In accordance with CBD, HELCOM's overall goal of a favourable conservation status of Baltic Sea biodiversity is described by the following three ecological objectives (including respective targets and indicators):

- natural marine and coastal landscapes;
- thriving and balanced communities of plants and animals; as well as
- viable populations of species.

In order to make the ecological objectives operational and to assess how the objectives have been achieved a number of initial targets and indicators have been developed.

The BSAP does acknowledge that the overall goal "favourable conservation status of the Baltic Sea biodiversity" cannot be reached without a broad consideration of human activities and needs for strong actions in other segments (maritime activities, eutrophication, and hazardous substances). To see objectives, targets and indicators for the other sections please refer to the BSAP document.

Nature conservation and biodiversity

Ecological objectives for nature conservation and biodiversity will be measured by the following initial indicators and targets. At this stage, it is not mentioned in the BSAP, when the preliminary set of indicators will be finalized, nor is it stated in the BSAP what exactly this process will look like.

In the section of 'nature conservation and biodiversity', most of the targets are not provided with concrete thresholds, indicated either by the wording (e.g., "close to", "largely", "sufficient") or because they are not specified. Also most of the indicators are formulated as trend indicators or as relative values ("percentage of").

Also it should be noted, that the indicators are not linked to a specific target, but just are listed under the same objective.

Objective: Natural marine and coastal landscapes

Targets:

- By 2010 to have an ecologically coherent and well-managed network of Baltic Sea Protected Areas (BSPAs), Natura 2000 areas and Emerald sites in the Baltic Sea;
- By 2012 to have common broad-scale spatial planning principles for protecting the marine environment and reconciling various interests concerning sustainable use of coastal and offshore areas, including the Coastal Strip as defined in HELCOM Rec. 15/1;
- By 2021 to ensure that "natural" and near-natural marine landscapes are adequately protected and the degraded areas will be restored.

Preliminary indicators:

- Designated BSPAs, Natura 2000 and Emerald site area as percentage of total subregion area;
- Percentage of important migration and wintering areas for birds within the Baltic Sea area which are covered by the BSPAs, Natura 2000 and Emerald sites;
- Percentage of marine and coastal landscapes in good ecological and favourable status;
- Percentage of endangered and threatened habitats/biotopes' surface covered by the BSPAs in comparison to their distribution in the Baltic Sea;
- Trends in spatial distributions of habitats within the Baltic Sea regions.

Objective: Thriving and balanced communities of plants and animals

Targets:

- By 2021, that the spatial distribution, abundance and quality of the characteristic habitat-forming species, specific for each Baltic Sea sub-region, extends close to its natural range;
- By 2010 to halt the degradation of threatened and/or declining marine biotopes/habitats in the Baltic Sea, and by 2021 to ensure that threatened and/or declining marine biotopes/habitats in the Baltic Sea have largely recovered;
- To prevent adverse alterations of the ecosystem by minimising, to the extent possible, new introductions of non-indigenous species.

Preliminary indicators:

- Percentage of all potentially suitable substrates covered by characteristic and healthy habitat-forming species such as bladderwrack, eelgrass, blue mussel and stoneworts;
- Trends in abundance and distribution of rare, threatened and/or declining marine and coastal biotopes/habitats included in the HELCOM lists of threatened and/or declining species and habitats of the Baltic Sea area;
- Trends in trophic structure and diversity of species (e.g., caught in scientific surveys);
- Trends in the numbers of detections of non-indigenous aquatic organisms introduced into the Baltic Sea.

Objective: Viable populations of species

Targets:

- By 2021 all elements of the marine food webs, to the extend that they are known, occur at natural and robust abundance and diversity;
- By 2015, improved conservation status of species included in the HELCOM lists of threatened and/or declining species and habitats of the Baltic Sea area, with the final target to reach and ensure favourable conservation status of all species;
- By 2012 spatial/temporal and permanent closures of fisheries of sufficient size/duration are established thorough the Baltic Sea area;
- By 2009, appropriate breeding and restocking activities for salmon and sea trout are developed and applied and therefore genetic variability of these species is ensured;
- By 2009 illegal, unregulated and unreported fisheries are close to zero;
- By 2008 successful eel migration from the Baltic Sea catchment area to the spawning grounds is ensured and national programmes for conservation of eel stocks are implemented;
- By 2015, as the short-term goal, to reach production of wild salmon at least 80%, or 50% for some very weak salmon river populations, of the best estimate of potential production, and within safe genetic limits, based on an inventory and classification of Baltic salmon rivers;
- By 2015, to achieve viable Baltic cod populations in their natural distribution area in Baltic proper;

- By 2015, to have the re-introduction programme for Baltic sturgeon in place, and as a long-term goal, after their successful re-introduction has been attained - to have best natural reproduction, and populations within safe genetic limits in each potential river;
- By 2015 bycatch of harbour porpoise, seals, water birds and non-target fish species has been significantly reduced with the aim to reach bycatch rates close to zero;
- By 2015 discards of fish are close to zero (<1%).

Preliminary indicators:

- Trends in the number of threatened and/or declining species;
- Abundance, trends and distribution of Baltic seal species compared to the safe biological limit (limit reference level) as defined by HELCOM HABITAT;
- Abundance, trends, and distribution of Baltic harbour porpoise;
- Number of rivers with viable populations of Baltic sturgeon;
- Spawning stock biomass of western Baltic cod and eastern Baltic cod compared to precautionary level (Bpa) as advised by ICES and/or defined by EC management plans;
- Fishing mortality level of western Baltic cod and eastern Baltic cod, compared to precautionary level (Fpa) as advised by ICES and/or defined by EC management plans;
- Trends in numbers of discards and bycatch of fish, marine mammals and water birds;
- Number of entangled and drowned marine mammals and water birds;
- Number of salmon rivers with viable stocks;
- Trends of salmon smolt production in wild salmon rivers.

7.3.2.2 Inventory of the provisions in the plans for management measures, policies, etc. in the Biodiversity Section of the BSAP

In relation to the Biodiversity Section of the BSAP, a large number of direct and indirect provisions are mentioned that reflect the BSAP goals and targets. These are summarised in the index below (Figure 7.3.2.2.1). The HELCOM Baltic Sea Action Plan Index outlines main actions in the BSAP, actors responsible for implementing the actions, lead Contracting Parties, relevant deadlines and remarks. The Index is be regularly updated with information from responsible actors on status of implementation of the BSAP. All the actions are numbered consequently and indexed in order to provide cross-reference to specific segments/paragraphs of the BSAP (as of 02.07.2009).

##	B BSAP §	III BIODIVERSITY AND NATURE CONSERVATION SEGMENT Reference to the HELCOM BSAP	Deadline	Main responsibility	Actions taken/planned	Remarks
38.1	B-1, B-2, B-3	Develop jointly broad-scale, cross-sectoral, marine spatial planning principles based on the ecosystem approach	2010	Contracting Parties All HELCOM Subsidiary Groups HELCOM SCALE (EU Project on spatial planning) Need for a Lead party: [Germany]	HELCOM Workshop on broad-scale marine spatial planning was organised on 27-29 January 2009 to draft the joint planning principles, HELCOM 30/2009 agreed that HELCOM will take initiative to develop a pilot project with other Baltic organisations for Maritime Spatial Planning in the Baltic to implement the EU Roadmap on Maritime Spatial Planning along with the HELCOM BSAP and Recommendation, HELCOM SCALE project ongoing	HELCOM SCALE project web page
38.2	B-1, B-2, B-3	Test, apply and evaluate broad-scale, cross-sectoral, marine spatial planning principles based on the ecosystem approach	2012	Contracting Parties All HELCOM Subsidiary Groups HELCOM SCALE (EU Project on spatial planning) Need for a Lead party: [Germany]	cf. 38.1	cf. 38.1
39.1	B-4	Designation of HELCOM Baltic Sea Protected Areas (BSPAs) from the already established MPAs	2009	Contracting States HELCOM HABITAT Lead party: Germany until 2010 with Secretariat support	Assessment of the status and ecological coherence of the MPA network is included in the Biodiversity assessment. HELCOM HABITAT 11/2009 agreed on a roadmap towards assessment of the status and ecological coherence of the MPA network which should be presented to the HELCOM 2010 Ministerial meeting to review progress in designations and implementation of management plans.	Biodiversity in the Baltic Sea - An integrated thematic assessment on biodiversity and nature conservation in the Baltic Sea (2009) (BSEP No. 116B)
39.2	B-4	Designation of HELCOM Baltic Sea Protected Areas (BSPAs) - new MPAs	2010	Contracting States HELCOM HABITAT Lead party: Germany until 2010 with Secretariat support	cf. 39.1	cf. 39.1
40	B-5.a	Assessment of ecological coherence of the BSPA/MPA network (Joint HELCOM/OSPAR working programme to the 2003 Ministerial Declaration)	2010	HELCOM HABITAT Lead: Secretariat with support from Germany	cf. 39.1	cf. 39.1
41	B-5.b	Finalisation and where possible implementation of management plans for Baltic Sea Protected Areas	2010	Contracting States HELCOM HABITAT Lead party: Germany	cf. 39.1	cf. 39.1
42	B-7.c	Further development of detailed landscape maps	2010	Contracting States HELCOM HABITAT Lead party: Sweden	The Biodiversity assessment includes a chapter on marine landscapes. Contracting States to further elaborate national landscape maps and to provide updated maps to the HELCOM Secretariat, HELCOM HABITAT 11/2009 to review progress	Biodiversity in the Baltic Sea - An integrated thematic assessment on biodiversity and nature conservation in the Baltic Sea (2009) (BSEP No. 116B) Landscape maps produced by the BALANCE project
43	B-7.a	Updating of a complete classification system for Baltic Sea marine habitats/biotopes	2011	Contracting Parties HELCOM HABITAT Lead party: Estonia [and Sweden]	HELCOM HABITAT 11/2009 agreed to link the updating of classification to the work of the HELCOM Red List project. First expert workshop to be organized in autumn 2009.	HELCOM Red List project web page
44	B-7.b	Updating of HELCOM Red lists of Baltic habitats/biotopes and biotope complexes	2013	Contracting Parties HELCOM RED LIST, HELCOM HABITAT Need for a Lead party: [Germany, Sweden and Finland]	HELCOM RED LIST project has started its work and will develop the List by 2013, HELCOM HABITAT 11/2009 to reviewed progress, first expert workshop to be held in autumn 2009.	This is liked to updating of a complete classification system for Baltic Sea marine habitats/biotopes cf. A 43 HELCOM Red List project web page
45	B-7.d	Identification and mapping of potential and actual habitats of habitat forming species (bladder wrack, eelgrass, blue mussel, stoneworts) and development of a common approach for the mitigation of negative impacts	2013	Contracting Parties HELCOM HABITAT Lead party: Sweden	Contracting States to act, Integrated thematic assessment of biodiversity and nature conservation Annex IV lists the existing models	Biodiversity in the Baltic Sea - An integrated thematic assessment on biodiversity and nature conservation in the Baltic Sea (2009) (BSEP No. 116B)
46	B-7.b	Producing a comprehensive HELCOM Red list of Baltic Sea species	2013	Contracting Parties HELCOM HABITAT Need for a Lead party: [Germany, Sweden and Finland]	HELCOM RED LIST project has started its work and will develop the List by 2013, First expert workshops of the species groups will be held in October 2009	HELCOM Red List project web page
47	B-7.e	Develop research on reintroduction of valuable phytobenthos species in regions of their historical occurrence		Contracting States HELCOM HABITAT Need for a Lead party: [Poland]	HELCOM HABITAT 11/2009 reviewed the progress with no major outcomes.	
48	B-7.f	Production of an assessment of the conservation status of non-commercial fish species	2011	Contracting States HELCOM FISH HELCOM HABITAT HELCOM MONAS Lead party: Sweden [Estonia]	HELCOM FISH project works on the issue, outcome in 2011, project to report to HELCOM HABITAT 11/2009	Outline for the Assessment can be found in the Minutes of the HELCOM FISH 2/2009
49	B-7.g	Further development of a coordinated reporting system and database on harbour porpoise sightings, by-catches and strandings	2010	Contracting States HELCOM HABITAT ASCOBANS, ICES Lead party: Finland	HELCOM SEAL to establish expert links to the Jastarnia Group/ ASCOBANS to make use of the work of the Jastarnia Group and to avoid duplication of work, possibilities for incorporating the porpoise database, developed by Baltic Sea porpoise project and now held in Forschungs-und-Technologiezentrum Westküste, Germany, to the HELCOM databases are being studied by the Secretariat	

50	B-7.h	Promotion of research on developing methods for assessing and reporting on impacts of fisheries on biodiversity		Contracting States HELCOM HABITAT Relevant cooperation organisations	HELCOM Contracting Parties' participation in BONUS+ the Baltic Sea Research Programme	
51	B-7.i	Development and implementation of effective monitoring and reporting systems for by-caught birds and mammals		Contracting States/fisheries authorities HELCOM HABITAT Need for a Lead party: [Finland]	HELCOM SEAL 3/2009 initiated reporting and compilation of data on by-caught mammals	Report of HELCOM SEAL 3/2009
52	B-8	Development and implementation of fisheries management measures for fisheries inside marine protected areas	2010	Contracting Parties/fisheries authorities HELCOM HABITAT Baltic RAC Lead party: Germany	Compilation of experiences for application of available management options from the ongoing and completed projects will be presented to the next Meeting of the Fisheries/Environmental Forum for Implementation of the HELCOM Baltic Sea Action Plan in September-October 2009	Relevant EC regulations shall be taken into account
53	B-9	Finalisation and implementation of national management plans and implementation of non-lethal mitigations measures for seals-fisheries interactions (HELCOM Recommendation 27-28/2)	2012	Contracting States HELCOM SEAL HELCOM HABITAT Lead Party: Sweden	Contracting States to act and *ad hoc* HELCOM SEAL expert group monitors progress in developing and implementing the national management plans and implementation of non-lethal mitigations measures for seals-fisheries interactions with reporting to HELCOM HABITAT group	Latest review on implementation of HELCOM Recommendation 27-28/2 was carried out by HELCOM SEAL 3/2009
54	B-10, B-11	Baltic Sea shall become a model of good management of human activities; all fisheries management be developed and implemented based on the Ecosystem Approach in order to enhance the balance between the sustainable use and protection of marine resources		Contracting Parties and Observers to HELCOM Need for a Lead party: EC [and Russia]	Contracting Parties to report on the actions taken by the competent authorities and Baltic Fisheries/Environmental Forum for Implementation of the HELCOM Baltic Sea Action Plan Fish/Fisheries related items to work on the issue	Relevant EC regulations shall be taken into account
55	B-12	The competent fisheries authorities to take all the necessary measures to ensure that populations of all commercially exploited fish species are within safe biological limits, reach Maximum Sustainable Yield, and are distributed through their natural range, and contain full size/age range	2021	Contracting Parties/fisheries authorities HELCOM HABITAT Baltic RAC Need for a Lead party: EC [and Russia]	Fisheries/Environmental Forum for Implementation of the HELCOM Baltic Sea Action Plan will keep this issue under constant review in line with elaboration of the joint input to EC Common Fisheries Policy	Relevant EC regulations shall be taken into account
56	B-13.a	Development of long-term management plans for commercially exploited fish species (salmon, sea trout, pelagic species and flatfish)	2010	Contracting Parties/fisheries authorities HELCOM HABITAT Baltic RAC Need for a Lead party: EC [and Russia]	Fisheries/Environmental Forum for Implementation of the HELCOM Baltic Sea Action Plan will keep this issue under constant review in line with elaboration of joint submission for EC Common Fisheries Policy in 2012	Relevant EC regulations shall be taken into account
57	B-13.b	Introduction of additional fisheries management measures to achieve: - that all caught species and by-catch are landed and reported - continued designation of additional/improved spatial and/or temporal closures - designation of additional permanent closures - further development and application in all cases of appropriate breeding and restocking practices for salmon and sea trout - minimisation of by-catch of under-sized fish and non-target species - an evaluation of the effectiveness of existing technical measures to minimise of by-catch of harbour porpoises and to introduce adequate new technologies and measures (by 2008)	2012	Contracting Parties/fisheries authorities HELCOM HABITAT Baltic RAC Need for a Lead party: EC [and Russia]	Fisheries/Environmental Forum for Implementation of the HELCOM Baltic Sea Action Plan will keep this issue under constant review and will continue information exchange cf. 52	Relevant EC regulations shall be taken into account
57.1	B-13.b	Evaluation of the effectiveness of existing technical measures to minimise of by-catch of harbour porpoises and to introduce adequate new technologies and measures	2008	Contracting Parties/fisheries authorities HELCOM HABITAT Baltic RAC Need for a Lead party: EC [and Russia]	Better information on harbour porpoise by-catch needed, e.g. through traditional and new catch control measures; Proposed Baltic-wide LIFE+ SAMBAH -project would contribute to the assessment of harbour porpoise population; Further information exchange will continue within the framework of Fisheries/Environmental Forum for Implementation of the HELCOM Baltic Sea Action Plan as well as within HELCOM SEAL Expert group	Evaluation of the effects of e.g. ban of drift nets, beginning on 1 January 2008 (EC Regulation 2187/2005) Relevant EC regulations shall be taken into account
58	B-14.a	Elimination of illegal, unregulated and unreported (IUU) fisheries and further development of landing control	immediately	Contracting Parties / fisheries authorities, HELCOM HABITAT. Need for a Lead party: EC [and Russia]	Fisheries/Environmental Forum for Implementation of the HELCOM Baltic Sea Action Plan will keep this issue under constant review	Relevant EC regulations shall be taken into account
59	B-14.b, B-15	Implementation of existing long-term management plans for cod and eel. The competent authorities to apply, in relation to the recommendation above, the targets annexed to the Action Plan	2012	Contracting Parties/fisheries authorities HELCOM HABITAT Lead party: Sweden	HELCOM Secretariat (Fisheries Project Researcher) 2008-2009 and Baltic Fisheries/Environmental Forum for Implementation of the HELCOM Baltic Sea Action Plan Fish/Fisheries related items to work on the issue Cf. 61.1	Relevant EC regulations shall be taken into account
60	B-16	A joint submission by EU Member States to the 2012 review of EU Common Fisheries Policy	2012	Contracting States that are also EU Member States HELCOM HABITAT	Fisheries/Environmental Forum for Implementation of the HELCOM Baltic Sea Action Plan will keep this issue under constant review given the timeline for the revision process as agreed by FISH/ENV FORUM 2/2008	

61.1	B-17	Additional fisheries measures such as national programmes for eel stocks	2008	Contracting Parties/fisheries authorities HELCOM HABITAT Lead party: Sweden	National management plans for eel were elaborated and submitted to EC for evaluation, implementation of the plans has started	Relevant EC regulations shall be taken into account
61.2	B-17	Additional fisheries measures such as classification and inventory of rivers with historic and existing migratory fish species	2012	Contracting Parties/fisheries authorities HELCOM HABITAT Lead party: Sweden	Cf. 61.4	Relevant EC regulations shall be taken into account
61.3	B-17	Additional fisheries measures such as development of restorations plans to reinstate migratory fish species	2010	Contracting Parties/fisheries authorities HELCOM HABITAT Lead party: Sweden	Fisheries/Environmental Forum for Implementation of the HELCOM Baltic Sea Action Plan will keep this issue under constant review and continue information exchange HELCOM FISH Project will also take this issue into account while elaborating assessment of the conservation status of non-commercial fish species	Relevant EC regulations shall be taken into account
61.4	B-17	Additional fisheries measures such as conservation of at least ten wild salmon river populations as well as the reintroduction of native salmon in at least four potential salmon rivers	2009	Contracting Parties/fisheries authorities HELCOM HABITAT Lead party: Sweden	HELCOM Secretariat has prepared questionnaire to be sent out to the Contracting Parties in order to make an inventory and classification of salmon and sea trout rivers, assess current status of wild salmon population and further decide on possible conservation measures; EC funding for this project was foreseen, but still pending	Relevant EC regulations shall be taken into account
62	B-21.a	Establish a cooperation network to agree on guidelines to promote the ecosystem-based management of coastal fisheries		Contracting States/fisheries authorities HELCOM HABITAT HELCOM FISH Lead party: Sweden [Estonia]	HELCOM FISH Project will address this issue through its work Ecosystem-based management of coastal fisheries was also addressed by Fisheries/Environmental Forum for Implementation of the HELCOM Baltic Sea Action Plan	
63	B-18	Enhance restoration of lost biodiversity by supporting German/Polish action to reintroduce Baltic sturgeon		Germany, Poland HELCOM HABITAT Lead party: Germany	Germany together with Poland is carrying out a reintroduction programme in Odra River basin and has developed and translated a reporting format for distribution to Baltic Sea countries. Those countries reported to HELCOM HABITAT	HELCOM HABITAT minutes
64	B-21.b, B-21.c	Development of long-term management plans and a suite of indicators for coastal fish species	2012	Contracting States/fisheries authorities HELCOM HABITAT HELCOM FISH Lead party: Sweden [Estonia]	HELCOM FISH project to develop long-term management plans and a suite of indicators, HELCOM Secretariat (Fisheries Project Researcher) 2008-2009 and Baltic Fisheries/Environmental Forum for Implementation of the HELCOM Baltic Sea Action Plan Fish/Fisheries related items to work on the issue	

Figure 7.3.2.2.1. The 'HELCOM Baltic Sea Action Plan Index' for the biodiversity section of the BSAP from July 2007, outlining main actions in the BSAP, factors responsible for implementing the actions, lead Contracting Parties, relevant deadlines and remarks.

7.4 Intersessional workplan

The original intention to "ground-truth" the objectives and management measures that are contained in the ecosystem plans against the science that was available to the system does not appear to be a promising pathway. The differences between the two plans examined in detail in this section so far highlight that the notion of an "integrated Ecosystem Management Plan" (IEMP – but going by many different names in different jurisdictions) covers a very wide range of types of plans. They can differ in the level of the objectives set – from aspirational and conceptual to quite specific and operational. They can differ in the degree to which they contain specific management provisions, and whether the provisions are oriented more at outcomes or at regulatory actions. This diversity is neither a strength nor a weakness of the IEMPs; just an inescapable consequence of the extended social and governance processes that are central to development of the plan. Those social and governance processes are always going to vary greatly from area to area: the Barents Sea Plan was a product of a single country; the Baltic Plan was the product of several countries coordinated through a formal regional seas organization. These governance and social differences are rooted in cultures, national laws and regional agreements and are not likely to converge soon. Therefore it is appropriate to plan for a continued diversity of contents in the category of IEMPs.

This inescapable diversity in the contents of IEMPs makes it unrealistic and probably unhelpful to pursue a line of evaluation that would suggest that there is some single "right" level of science input to IEMPs, or even some single "right" degree of linkage between the plans and the science available for their development. However, our review revealed other pathways to explore and provide constructive guidance for the relationship between science and the development of the IEMPs. These pathways build on some of the positive conclusions that also came from our consideration of the IEMPs. One of those positive conclusions was that whatever the social processes may or may not have done with the science, they certainly contributed to widespread stakeholder buy-in to the final products. This provides a much more solid basis for any future efforts at improved management (integrated or not). The processes have also resulted in greater cooperation among various government departments and levels of government, which has positive implications for likelihood of some success at more integrated management the future. In this context we propose several follow-up activities for the following meeting of WGECO, and intersessionally between now and then.

1) **Information availability and objectives setting**: How does the availability of science information interact with the ability to develop general or specific objectives and management measures for IEMPs? What made it possible to have the numerous specific measures in the Baltic Plan? Were the high level ecosystem objectives in the Barents Sea Plan dependent on the number of ecosystem overviews that were prepared? The intent of this review /discussion would be to inform governance processes of the scale of investment in science that is necessary to deliver different scales and specificities of objectives and management measures. It may also help to manage expectations for the level of specificity in objectives and management measures that can be realized, given an idea of the amount and types of information that could be made available.

2) **Importance of science support in different stages of IEMP development**: Science support is needed both to provide the information on which to initiate the process of developing IEMPs, and throughout that process, as participants may pose new questions or want to pursue certain issues in greater depth. Reviewing and summarizing how science demands were met before, during, and after development of the IEMPs could be an informative direction to take. Are there practices that increase the burden on science to support their development, that decrease the burden, and at least increase value of the science that was input to the process?

Development of the IEMPs that were reviewed herein is only one stage in a process of more integrated management. Implementation lies ahead for the Baltic (and other) plans, and the Barents Sea Plan was implemented only four years ago. With regards to science support for the next steps in the process we propose:

3) **Importance of inter-agency communication for IEMP implementation efficiency:** Implementation of the IEMPs will require coordinated action by many agencies, with differences in mandates, priorities, risk tolerances, and operational cultures. What challenges does this pose for interagency (and often international) cooperation? Is it possible to cross-tabulate the objectives and/or measures in the plan with the agencies (including civil society) that would have to cooperate? Are there differences in how these agencies use science information and advice that might impede progress? What constructive roles can science play in increasing the likelihood of effective interagency and (international) cooperation?

4) **Criteria for evaluating IEMPs**: At some point the consequences of IEMPs will have to be evaluated; one evaluation has already been completed for the Barents Sea Plan. The evaluations are likely to include social and economic consequences outside ICES traditional scope of activities. However

the ecological consequences will have to be evaluated, likely both with regard to progress towards objectives and the scale of benefits relative to scale of costs. How should such evaluations be conducted while keeping the workload on the science community within bounds? For example, what would be criteria for success? What information would be needed and how would it be best used?

A review next year around these general questions could contribute to more efficient use of science in the development and implementation of IEMPs, and to IEMPs being both stronger and more likely to produce the desired benefits. Intersessionally experts involved in both the Barents Sea and Baltic Sea Plans will undertake preparations to be ready to discuss these questions, with as much relevant documentation as can be assembled. Efforts will be made to attract participants who have experience with supporting development and implementation of other IEMPs, such as possibly those for Puget Sound, Washington, the Canadian Scotian Shelf, and Chesapeake Bay, Maryland/Virginia.

7.5 References

Anon. 2006. Report No. 8 to the Storting (2005–2006) Integrated Management Plan of the marine Environment of the Barents Sea and the Sea off the Lofoten Islands. The Royal Ministry of the Environment, Oslo 2006. 144 pp.

Backer H. 2008. Indicators and scientific knowledge in regional Baltic Sea. Environmental policy. ICES J. Mar. Sci. 65: 1398–1401.

Backer H, Leppänen J-M, Brusendorff AC, Forsius K, Stankiewicz M, Mehtonen J, Pyhälä M, Laamanen M, Paulomäki H, Vlasov N, Haaranen T. In press. HELCOM Baltic Sea Action Plan – A regional programme of measures for the marine environment based on the Ecosystem Approach. Mar. Pollut. Bull.

Backer H, Leppänen J-M. 2008. The HELCOM system of a vision, strategic goals and ecological objectives: implementing an ecosystem approach to the management of human activities in the Baltic Sea. Aquatic Conservation: Freshwater and Marine Ecosystems 18, 321–334.

Bjørge, A., Borge, A., Kleven, S. 2006a. Observed and reported bycatches of marine mammals in Norwegian shelf and offshore fisheries. NAMMCO/15/MC/BC/7. 9 pp.

Bjørge, A., Godøy, H., Nedreaas, K. 2006b. A system for monitoring bycatches of marine mammals in Norwegian coastal and inshore waters. NAMMCO/15/MC/BC/8. 9 pp.

Bjørge, A., Hartvedt, S., Ynnesdal, H. 2006c. Spatial structure of Norwegian fisheries and the associated risk for bycatches of marine mammals. NAMMCO/15/MC/BC/6. 9 pp.

FAO. 2005. Putting into Practice the Ecosystem Approach to Fisheries. FAO, Rome.

Garcia, S. M., Zerbi, A., Aliaume, C., Do Chi, T., Lasserre, G. 2003. The ecosystem approach to fisheries. Issues, terminology, principles, institutional foundations, implementation and outlook. FAO Technical Paper, 443. 71 pp.

HELCOM, 1974. Convention on the Protection of the Marine Environment of the Baltic Sea Area.

HELCOM. 2006. Development of Tools for Assessment of Eutrophication in the Baltic Sea. Baltic Sea Environment Proceedings 104.

HELCOM. 2007a. HELCOM Baltic Sea Action Plan (adopted by the HELCOM Ministerial meeting, Krakow, Poland 15th November 2007).

HELCOM. 2007b. Towards favourable conservation status of Baltic Sea biodiversity. HELCOM Overview 2007, HELCOM Ministerial Meeting, Krakow, Poland, 15 November 2007.

HELCOM. 2009a. Eutrophication in the Baltic Sea – an integrated thematic assessment of the effects of nutrient enrichment in the Baltic Sea region. Baltic Sea Environment Proceedings 115B.

HELCOM. 2009b. Biodiversity in the Baltic Sea – an integrated thematic assessment on biodiversity and nature conservation in the Baltic Sea. Baltic Sea Environment Proceedings 116B.

O'Boyle, R., Jamieson, G. 2006. Observations on the implementation of ecosystem-based management: experiences on Canada's east and west coasts. Fisheries Research, 79: 1–12.

O'Boyle, R., Sinclair, M., Keizer, P., Lee, K., Ricard, D., Yeats, P. 2005. Indicators for ecosystem-based management on the Scotian Shelf: bridging the gap between theory and practice. ICES Journal of Marine Science, 62: 598–605.

Olsen E., Gjøsæter, H., Røttingen, I., Dommasnes, A., Fossum, P., Sandberg, P. 2007. The Norwegian ecosystem-based management plan for the Barent Sea. Short Communication. ICES Journal of Marine Science, 599–602.

Ridgeway, L., Maquieira, C. 2006. Report on the work of the United Nations Open-ended Informal Consultative Process on Oceans and the Law of the Sea at its 7th meeting. 28 pp.

Stiansen, J.E., Korneev, O., Titov, O., Arneberg, P. (Eds.), Filin, A., Hansen, J.R., Høines, Å., Marasaev, S. (Co-eds.). 2009. Joint Norwegian-Russian environmental status 2008. Report on the Barents Sea Ecosystem. Part II – Complete report. IMR/PINRO Joint Report Series, 2009(3), 375 pp. ISSN 1502–8828.

Sunnanå, K, Fossheim, M, Olseng, C. (eds). 2010. Forvaltningsplanen Barentshavet-rapport fra Overvåkingsgruppen 2010. Fisken og Havet, særnr. 1b-2010 (in Norwegian).

von Quillfeldt, C., and Dommasnes, A., (eds.). 2005. Report of the working group on fish stocks and fisheries (including production and energy flows). The Scientific Basis for Environmental Quality Objectives (EcoQOs) for the Barents Sea Ecosystem. Norway, 2005 170 pp. (in Norwegian).

8 ToR f) Extending marine assessment and monitoring framework used in Chapter 10 of the QSR 2010 (OSPAR request 2010/1)

f) Extending marine assessment and monitoring framework used in Chapter 10 of the QSR 2010 (OSPAR request 2010/1)

To review the methodology used by the OSPAR workshop on the development of Chapter 11 of the QSR 2010 (Utrecht workshop) and taking into account, inter alia, ICES work on integrated assessment, provide advice on the following aspects:

i) improvements that could be made to the thresholds between different assessment classes, including any scientific basis for proposed thresholds;

ii) extending the methodology to support the assessment of plankton communities; (Utrecht workshop);

iii) improving the method for working at different scales, such as the level of an OSPAR Region, the level of sub-Regions such as the Irish Sea or the Channel or the level of an estuary or an MPA.

8.1 Introduction

The aim of the assessment was to assess the status of key ecosystem components in the OSPAR Regions, within the wider context of the OSPAR QSR 2010. The process was described in detail in Robinson *et al*. (2009), and the development of the approach in Robinson *et al*. (2008). There are many examples of methods to assess threat or risk of impact of particular activities, including well established risk assessment frameworks. In most cases, however, these are for either single (Tyler-Walters *et al*., 2007) or multiple pressures (Halpern *et al*., 2007), on one type of marine component (Carlin and Rogers, 2002), or for single pressures on multiple components (Fletcher, 2005). Where they cover multiple pressure/component interactions, the assessments are usually done independently for each pressure/component interaction resulting in a potential lack of consistency between them (OSPAR, 2003; JNCC, 2007). There have been other attempts to develop integrated approaches, for example, REGNS (Kenny *et al*., 2009), and the Australian 3-tier ecosystem risk assessment framework (Smith *et al*., 2007), as well as research programs undertaking comparative evaluations of threats to ecosystems, for example, the IndiSeas project (Shin and Shannon, 2010). The aim in the OSPAR QSR assessment was to try to simultaneously assess the importance of different pressures across multiple components in a number of very different marine ecosystems. The process was designed to use coherent definitions and, particularly, thresholds between classes of response (i.e., good, moderate or poor) to provide consistency between the ecosystem areas and components.

8.2 Provide advice on improvements that could be made to the thresholds between different assessment classes, including any scientific basis for proposed thresholds

8.2.1 Overview of OSPAR QSR approach

The setting of thresholds between categories of impact is central to the methodology. Robinson *et al*. (2008) described the difficulty in selecting thresholds for state indicators that are scientifically justified (i.e., based on a robust relationship between the level of perturbation and recovery potential of ecosystem components). The thresholds should also be relevant to the objectives of the assessment being undertaken. In the OSPAR assessment this was to assess status relative to former natural conditions. There were no clear guidelines available on how good status should be defined and as a result, the Robinson *et al*. (2009) methodology (used in Utrecht) used thresholds that were where possible based on regulations, e.g., Habitats Directive, with the rationale that they reflect agreements among States on desired levels of protection. However, the regulatory benchmarks do not cover all components of ecosystems. Extrapolating regulatory benchmarks widely to ecosystem components would require assuming both that society had equal levels of risk aversion to all ecosystem components, which is not the case (Rice and Legace, 2007), and that the sustainable level of impact on all ecosystem components was the same, which is also not the case (see below). In the case of OSPAR/Utrecht the species thresholds in particular were judged to lack scientific credibility and to be contested by stakeholders. An example of the thresholds is presented in Table 8.2.1.1.

Table 8.2.1.1. Criteria for used for species status in the OSPA QSR (taken from Robinson *et al*., 2009).

Table A3.1 Criteria used to assess the current status of species group components relative to former natural conditions. Descriptors apply to the aggregate view of a component.

	Status			
Threshold descriptor	Good	Moderate	Poor	Confidence
(i) Range	<10% of species have range declines >10% compared to former natural conditions.	10-50% of species have range declines >10% compared to former natural conditions	>50% currently have range declines >10% compared to former natural conditions.	*Low or High*
(ii) Population size (extent)	<10% of species currently have a large decline in population size (>25% relative to former natural conditions)	10-50% of species currently have a large decline in population size (>25% relative to former natural conditions)	>50% of species currently have a large decline in population size (>25% relative to former natural conditions)	*Low or High*
(iii) Population condition	<10% of species have strong deviations in reproduction, mortality or age structure relative to former natural conditions[1]	10-50% of species have strong deviations in reproduction, mortality or age structure relative to former natural conditions[1]	>50% of species have strong deviations in reproduction, mortality or age structure relative to former natural conditions[1]	*Low or High*
Current status	All 'green'	One or more 'amber' but no 'red'	One or more 'red'	Overall confidence Very low = 3/3 'Low' Low = 2/3 'Low' Moderate = 2/3 'High' High = 3/3 'High'

[1]Trend information required for clear deviation in reproduction, mortality or age structure showing a significant deviation from former natural conditions.

WGECO (ICES, 2009a) noted that there are some cases where scientifically derived reference points exist (e.g., B_{lim}, B_{pa} for assessed commercial fish stocks, harbour porpoise bycatch EcoQO) and that these could be used in any future assessment as a robust threshold. However, benchmarks like B_{lim}, and B_{pa} vary across species for sound biological reasons, and ICES is currently in the process of defining the basis for its fisheries management advice, which may result in changes in these benchmarks for many stocks. Issues surrounding the use of such reference points are discussed below in Section 8.2.2. For all the components, or parts of components (e.g., non assessed fish stocks) where such indicators are not currently available a large-scale data analysis exercise is required to define the relationships between levels of perturbation and either recovery potential, or impact in terms of loss of ecosystem structure, function, process or socio-economic benefits (see below). In the case of the MSFD, status should be assessed for ecosystem components relative to good environmental status where sustainable use of the ecosystem is achieved. As such, the MSFD itself does not define what thresholds correspond to sustainable use. However, the standards set in Agenda 21 of the Rio Declaration mean that the benchmark for sustainability impacts must be set above the point where there is serious risk of irreversible harm to components. This is discussed further in Section 6 of this Report.

8.2.2 Scientifically robust thresholds between different assessment classes (example for the fish community)

The following discussion is included solely as an illustration of what *might* be done in setting reference levels in the context of fish stocks. It should not to be taken as recommendation for any such use.

For individual ecosystem components, there are many examples of well worked out thresholds for a given indicator. The most obvious example would be the development of the Precautionary Approach (PA) reference points for assessed fish stocks. B_{lim} can be taken as the biomass level below which recruitment is impaired or the dynamics of the stock are unknown. B_{pa} functions as a tool to manage risk of falling below B_{lim}, given uncertainties in assessments and management. Hence it can be taken to be the biomass level above which the stock should be maintained to ensure a low risk to recruitment impairment, i.e,. of reaching B_{lim}. So one possibility would be that if a stock is over B_{pa} it could be considered to have a "good" status, between B_{pa} and B_{lim} it could be considered to have a "moderate" status, and below B_{lim} it could be considered to have a "poor" status. If we had access to MSY defined biomass levels (B_{msy}) and information on the variation in a stock around B_{msy} due to recruitment variation, then the lower boundary for "good" could be redefined as within a given probability range of B_{msy} , "moderate" between that biomass and B_{lim}, and "poor" below B_{lim}. The problem becomes more difficult when one wishes to extend this to *all* fish stocks, as was attempted for the fish ecosystem component in the OSPAR QSR. So for assessed fish stocks for a given area one could evaluate in relation to B_{pa} and B_{lim}, and produce a catalogue of those evaluations. This could indicate the proportion of stocks above B_{pa}, between B_{pa} and B_{lim}, and below B_{lim}. Even if one could assume that all stocks were assessed, there are no clear guidelines for how to interpret the number of stocks in each category as indicating good, moderate or bad status for the *ensemble*. (see Section 6.3.3). Additionally, the requirement of the OSPAR QSR was to establish changes in relation to "former natural conditions". Even if all stocks were to be over B_{pa}, it would still not establish their deviation from "former natural conditions", as B_{pa} is intended to express low risk of collapse rather than any long-term "typical" value. Most fish species in any given OSPAR region will not have analytical assessments carried out. It might be assumed that if all assessed stocks are considered as over B_{pa} then all other fish stocks would be expected to be in a similar state. However, this would only be the case if three assumptions were true:

a) fishing was the dominant source of human-induced mortality on all stocks;

b) no non-commercial fish stock had a higher catchability than commercial stocks and;

c) no non-commercial fish stock had a lower sustainable fisheries (bycatch) mortality rate than commercial stocks.

In addition, it is not necessarily clear whether one can assume that if all assessed stocks are below B_{pa}, then all other fish species are similarly impacted. It should be possible for some non-assessed stocks to be in a "poor" status, even if most assessed stocks were over B_{pa}. Most fisheries are targeted to some extent, and full coincidence of impact on assessed and non assessed stocks would not be expected. Equally, pressures other than those from fishing may well act on different species in different ways, e.g., rising temperatures. Data for species range and condition may be available in many cases from research surveys, but generally there would be no analogue to B_{pa} for these indicators.

The reason for labouring the discussion of the situation for the fish component is that in this case we have access to detailed, consistent and scientifically established stock level indicators in B_{pa} and B_{lim}, as well as detailed survey data. Despite this we currently do not have an accepted scientific solution for ranking the *ensemble* using this information. The approach taken for the OSPAR QSR at the species level, defined "good" population size as <10% of species currently have a large decline in population size (>25% relative to former natural conditions). These thresholds were adapted from "Favourable Conservation Status Criteria" used in the Habitats Directive assessments. For the fish community, it would be possible to replace the ">25%" condition, with a "below B_{pa}" condition, which would have a clearer scientific basis. The "10% of species" threshold value would still not be a scientifically derived threshold. Additionally, as discussed above, B_{pa} is NOT actually a measure of good status, but of low risk of impaired productivity, which, if continued would result in collapse.

8.2.3 Scientifically robust thresholds between different assessment classes (other components)

Most of the other species components do not have the same quality of information available, when compared to the fish community. In some components there will have been some information on abundance trends, range and condition on individual species, but again, no established scientific basis for evaluating the *ensemble* information. Habitat is a special case as unlike a species it generally has no real recoverability in ecological time (although there will be exceptions, e.g., biogenic habitats, although even these may take decades to centuries to recover). For the habitat components, there may be some information indicating changes in range, extent and damage to rock and biogenic reef habitats, for example see discussion on coral in the WGHAME report Section 8.3.2 (ICES, 2009b). But it would be difficult to assess this comprehensively due to the patchy and point sample nature of the data (Fosså *et al.*, 2002; Rice *et al.*, 2010). One alternative for habitats is to use pressure indicators and assess an area subject to pressure and some understanding of how the pressure affects condition or even extent of the habitat.

Other habitat components are less well described (see Section 4). In general there are no scientifically established thresholds established, and in the 2010 QSR assessment the thresholds were derived from the Habitats Directive (JNCC, 2007), and the OSPAR Texel-Faial Criteria (OSPAR, 2003).

8.2.4 Former natural conditions-constraint on reasonable use of data

A requirement in the OSPAR QSR approach was to evaluate against "*former natural conditions*". It was often difficult to establish these. For instance at what point in time were conditions "natural". In the North and Baltic Seas, fishing has been going on for centuries, especially for herring, and the fisheries and stocks are known to have undergone considerable natural, and human influenced changes (Awebro *et al.*, 2007; Poulsen *et al.*, 2007; Poulsen, 2010; Desse and Desse-Berset, 1993). Similar arguments could be made for benthic habitats in these areas (Robinson and Frid, 2008). Using this "former natural conditions" constraint made the setting of meaningful thresholds even more difficult in computational terms. It is also important that OSPAR should indicate how to understand what period they mean by "*former*". For a discussion of "pristine" conditions see Section 6.3.2.

8.2.5 WGECO approaches to defining thresholds

The following has been extracted from more extensive documentation provided in Sections 4 and 6 of this Report. These sections provide the supporting information underpinning these recommendations. It is important to understand that setting thresholds or reference levels is only one part of a larger process of scoping, determination of objectives, and determining the important pressures and impacts, and indicators for these. Only then, can we consider setting reference levels.

8.2.5.1 Stepwise process for identifying ecosystem components, and pressure and state indicators

WGECO consider that the following steps are required to ensure consistency within any process that would meet the requirements of the MSFD, or other integrated approaches:

1) An evaluation of the components of each regional ecosystem with regard to its "*structure, function and processes*", taking account of "*natural physiographic, geographic, biological, geological and climatic factors*" which identifies the parts of that particular ecosystem that are most crucial to its ecological integrity, structure, and function. In selecting these, indicators that relate to integrated aspects of the ecosystem (e.g., those that represent food web structure) should also be considered in order to capture the interactions of components within the regional ecosystem being assessed.

2) An evaluation of the major human activities that are likely to result in pressures in each regional ecosystem (including physical, acoustic, chemical and biological pressures), which identifies the pressures likely to be causing the greatest perturbations within that ecosystem, and the scales on which those pressures are operating.

3) Use of a scientifically peer-reviewed framework that consists of a cross-tabulation of pressure – ecosystem component interactions that reflects which types of ecosystem components are likely to be most impacted, or otherwise be most sensitive to the pressures identified in 2, and the pressures most likely to impact detrimentally the ecosystem components identified in 1. This cross-tabulation must also link back to the potential sources of pressures (e.g., the activity-pressure relationships identified in 2).

4) For the components and pressures that are evaluated to be most important, ensure that one or more robust and sensitive indicators are selected. Give particular attention to the interactions between the more important components from 1 and the more severe pressures from 2, which come out of the consideration in 3.

The consistency is, therefore, achieved by the functional equivalence of the indicators and reference levels. Indicators can be considered functionally equivalent when they are appropriate for measuring status of a pressure, structural or functional property or process that is of similar ecological significance across ecosystems, even if the exact indicators or properties differ across ecosystems.

8.2.5.2 Reference levels

The reference levels chosen should be representative of the objectives of the "management plan". In the OSPAR QSR, the objective was defined as "*former natural conditions*". When this guidance is taken as equivalent to "*pristine*", guidance for practice is presented in Section 6.3.2. The equivalence of "*former natural conditions*" and "*pristine*" is reasonable in some restricted cases, e.g., anthropogenic contaminants in water, and in those cases the benchmark is directly apparent as absence of the contaminants. However in many cases even if "*former natural conditions*" is interpreted as "*pristine*" the natural condition was non-zero and variable, and appropriate reference levels can be very difficult to define:

- With sufficiently long data-series we may have information prior to human impacts, but this is unlikely;
- If the ecosystem attribute is relatively stationary in space, except when impacted by human pressures, we could use an example from a reference unimpacted area. In practice this will likely be difficult, ecological attributes are rarely spatially stationary;
- We could use some form of process based model to project back to a time when the system was unimpacted. We would need to be very sure of the models and assumptions to do this;
- We could use statistical models based on observed data, provided there is sufficient contrast in the data (i.e., range of conditions). By definition we would only be doing the two steps above when lacking data of the "pristine" condition, so our models would be extrapolating, and this is where models are weakest;
- We could use expert opinion, as the last resort.

In addition, we would be assuming that "former natural conditions" could be replicated now and in the future. Even ignoring climate change, there are decadal and longer oscillations in the environment that could complicate the development of any approach based on former conditions.

Reference levels based on some equivalent of "sustainable use" or aiming at Good Environmental Status would a suitable alternative. They would also be more relevant to the international conventions of achieving sustainable use of our ecosystems (e.g., CBD). A reference point for sustainable perturbation might be derived by examining the relationship between perturbation level and recovery capacity. When this relationship is non-linear, a point where its slope changes most rapidly might provide an appropriate reference point. When this relationship is linear or non-defined, or when ecosystem attributes have no capacity to recover from perturbation, then the same kind of analysis has to be performed with the ecosystem function served by this attribute instead of its recovery capacity (see Section 4 of this report). In cases where the relationship between a perturbation and recovery capacity or ecosystem function is lacking or non-conclusive, a strategy that is appropriate to choose a reference level, for that system, reflects:

- For state indicators, the value of the indicator at a time when pressures affecting the indicator were considered sustainable;

- For pressure indicators, the value of the indicator from a time when the ecosystem components most sensitive to the pressure were considered to be in a GES state;
- If data are insufficient for the first two alternatives, the value of either type of indicator when scientifically sound analyses of historical data suggests that there is low likelihood that the structure, function or process represented by the indicator was not GES;
- If data are insufficient for the first three alternatives, the value of either type of indicator, at which theoretical or generic modelling results suggests that there is low likelihood that the structure, function or process represented by the indicator would be impaired;
- If data are still insufficient, evaluate in terms of trends, and possibly rate of trend, is the situation improving/deteriorating, and how fast. Establish threshold values for duration or amplitude of trend (Jennings and Dulvy, 2005).

As for indicators, reference levels can be considered functionally equivalent if they reflect the same level of sustainability, or risk of serious harm, across ecosystems even if the value of a given indicator (needed to be not at risk of harm or impacted unsustainably) varies across ecosystems (see discussion in Sections 4 and 6 of this Report).

8.2.5.3 Combining information across indicators for an overall assessment of "good environmental status"

One of the main concerns about the OSPAR QSR process was on the problems of integrating many indicators across component and ecoregion, and then setting percentage change criteria for "good, moderate and poor" status. Section 6.3.2 provides a detailed examination of the approaches need to combine multiple indicators to provide an overall assessment of GES, and the salient details are presented here. The issues would be to provide an adequate synthesis, that was useful, but that did not obscure important individual issues. For the MSFD several scales of need for combination were identified:

- Indicators within individual Attributes of a Descriptor (for complex Descriptors),
- Status across all the Attributes within a Descriptor,
- Status across all Descriptors.

The last of these is likely to be the most complex, and also the one that would be needed to develop an "overall assessment".

A number of methods were reviewed, none of which would fully satisfy the requirements. These are summarized in Table 8.2.5.3.1 extracted from Section 6.3.2.

A number of analytical approaches are under development that would address some of these issues and these are summarized in Sections 6.3.3.1–3.

In conclusion, at present there is no perfect method for combining and integrating indicators, but several promising avenues are being developed.

Table 8.2.5.3.1. Strengths and weaknesses of various integration methods, with a focus on characteristics particularly relevant to the MSFD (see Section 3.3): Pressure-State link (P-S link), and consistency and comparability (C-C).

METHOD	STRENGTHS	WEAKNESSES	P-S LINK	C-C
Kites, pie diagrams	Simple and transparent	Does not accommodate large indicator suites; ± prone to manipulation	No	No
Averages	Simple and understandable	Eclipsing; weighting issues	No	No
Composite indices		Eclipsing; not transparent	No	Yes
Multivariate methods (canonival correlation…)	Reduces redundancy, accounts for uncertainty	Data greedy	Exploratory and correlative	Probably not
Traffic light presentation	Easy to understand	Data greedy; weighting issues	Visual	No
Model-based methods (in development)	Rely on a consistent set of interrelated indicators	Complex models not transparent; complex output may need summarizing	Explicity assumed	Potentially

8.2.5.4 Thresholds between different assessment classes, including any scientific basis for proposed thresholds, in the OSPAR QSR approach

The Robinson *et al*. (2009) methodology (used in Utrecht) used thresholds that were generally based on existing regulatory limits (e.g., Habitats Directive), rather than based on any rigorous scientific rationale. The text provided above would indicate that at present, there is no complete or agreed methodology yet available to provide such a rationale, although the pieces of such a methodology are being developed. The indicators and associated thresholds generally have a scientific basis, e.g., GES. However, a scientifically robust way of establishing reference values over components or ecoregions is still under development.

The use of multiple levels in the OSPAR QSR (good, moderate and bad) made this process even less robust, as levels had to be set for this partition. WGECO takes the view that any use of threshold values can be counter productive. Ecological status is probably best viewed as a continuum, with GES at one end of that continuum. Although this requires some reference level above which we have achieved GES, this should not be used to suggest that all levels below any GES reference are equally acceptable. To put this into perspective, for a given species, such an approach would lead to situations where a species was close to extirpation being treated the same as if the species was, say, between B_{pa} and B_{lim}.

8.2.6 Conclusion

In response to the initial question in the ToR, if an integrated assessment such as the OSPAR QSR was to be the science basis for implementation of the MSFD, it would need to include:

- An explicit description of the relationships between pressure and state;
- A common approach that will ensure consistency and comparability across marine regions;

- Include multiple impacts and socio-economic aspects (e.g., in the context of the MSFD);
- Include consideration of data uncertainty or knowledge gaps.

It would need to follow the process outlined briefly in this section, and more extensively in Sections 3, 4 and 6. This can and should be done within the domain of the MFSD, and the results of the process should significantly improve any future QSR approach. Given that no clearly better way of evaluating the overall status of an eco-region or component exists thus far, the actual value of the thresholds used in the OSPAR QSR process is a matter of judgement alone. The issue is not really about the thresholds *per se* but about defining a robust process to evaluate the pressures that impact on ecosystem structure, function and processes, and to integrate these in a consistent fashion. This has not been achieved to date. This will require continued work by the scientific community. In addition, clearer policy objectives than "natural historical conditions," would greatly facilitate setting thresholds and choosing indicators for assessing Ecological Quality Status.

8.3 Extending the methodology to support the assessment of plankton communities; (Utrecht workshop)

WGECO were asked to consider whether it is possible to extend the methodology to support the assessment of plankton communities. This suggestion is supported by the report from the Meeting of the Management Group for the QSR ((MAQ)MAQ(2) 09/2/10 Add.3-E), and also by the ICES Working Group on Holistic Assessments of Regional Marine Ecosystems (WGHAME, ICES, 2009b). Inclusion of the plankton community would clearly enhance the holistic and integrative nature of the OSPAR assessment (see Section 3 of this Report), but would require changes to some aspects of the methodology.

8.3.1 Rationale for inclusion

1) A truly integrated ecosystem assessment should include state indicators of all key ecosystem components and, in particular, aspects of the ecosystem that are important to its structure and function (Section 3 of this Report). Changes in the structure and function of plankton (phyto- and zoo-, and mero-plankton) are of clear importance to the dynamics of many of the other components within the assessment, and support key ecosystem functions themselves (primary and secondary production).

2) Plankton data can provide some of the pressure information required by the assessment framework (and these may have already been considered by participants at the Utrecht workshop in describing the pressures). For example, one of the indices proposed by WGHAME would be a CPR derived invasive species index, and this could be used to provide information for the pressure "introduction of non indigenous species and translocations" in the process.

3) A further rationale could be that there exists good quality data on plankton communities from the Continuous Plankton Recorder (CPR) programme. Given the reasonably comprehensive spatial and excellent temporal coverage (see Figures 8.3.2.1), this could be regarded as one of the best datasets to make this sort of evaluation. However, WGECO have warned against the selection of ecosystem state variables where this is simply on the basis that "the data exist". In Section 3 of this Report we give guidance on how to select a comprehensive list of indicators that could be used in a fully in-

tegrative ecosystem assessment and this should be used to select the minimum number that is required for the plankton components. In the WGHAME report (ICES, 2009b), a series of possible CPR derived indicators is listed and the process described in Section 3 could be used to select any of those listed by WGHAME that would contribute to a scientifically robust integrated ecosystem assessment for a future OSPAR QSR.

8.3.2 Extension of methodology to include plankton

Plankton species, or community-based indices of plankton, as ecosystem components, would be best considered alongside the "species" level components; seabirds, cetaceans, seals and fish, of the Utrecht assessment. As discussed above the criteria and thresholds used in the Utrecht workshop were not appropriate for some of these components and this also applies for the plankton. In Section 3.5 of this Report, a general process is described for selection of indicators to represent state variables (criteria as described in the Utrecht methodology) and reference levels (thresholds as described in the Utrecht methodology) in relation to an assessment of status. This process should be applied to the plankton in further development of the Utrecht methodology. However, there are some points specific to the plankton that must be considered:

The criteria used to set thresholds between Good, Moderate and Poor status for the OSPAR QSR were based on declines in population range and size, and alterations in condition such that there were "strong deviations in reproduction, mortality or age structure relative to former natural conditions". The changes documented in Figure 8.3.2.1 include both substantial declines *and* substantial increases in abundance and ranges of different zooplankton assemblages in specific OSPAR regions. For example, sub-arctic species have declined in the North Sea, while temperate pseudo-oceanic species have increased substantially. The OSPAR assessment approach is generally predicated on impacts leading to "declines". For at least some of the plankton community indicators, pressures, e.g., warming, have led to decline in one group and increase in another. As another example, phytoplankton biomass and productivity (Edwards *et al*., 2009), have risen in most areas in recent years (Figure 8.3.2.2), due presumably to warmer conditions, and a longer growing season. This highlights the need to consider the "extent of change" for some state variables when setting reference levels, rather than a level that represents a difference in any particular direction.

The other issue to be clarified would be the reference to "former natural conditions". This issue also caused problems for some of the other components where "former natural conditions" were not clearly defined. Plankton communities undergo considerable amounts of natural variation in response to abiotic drivers. This must be taken into account when trying to choose an appropriate baseline for any assessment of status for this aspect of the ecosystem.

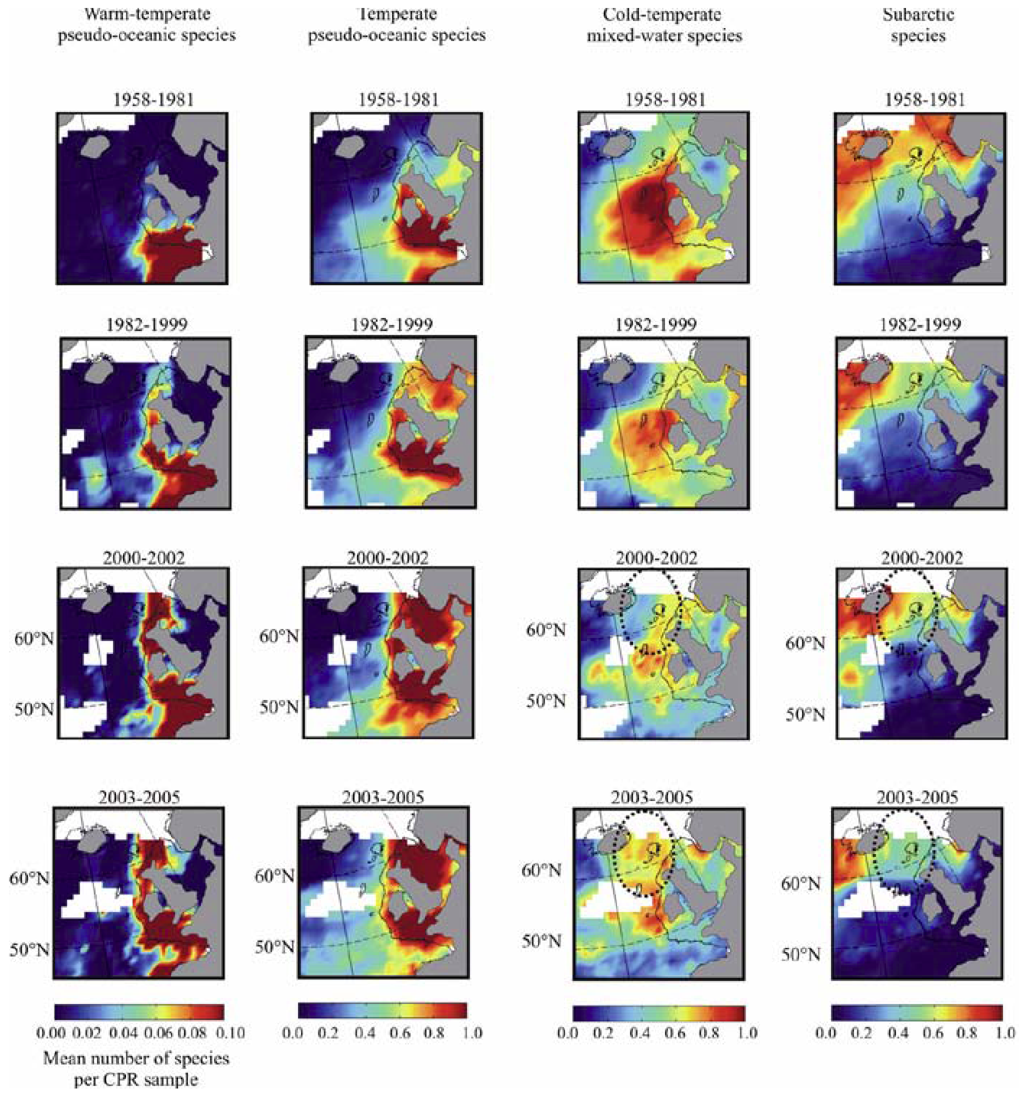


Figure 8.3.2.1. Biogeographical changes in four plankton assemblages spanning five decades. Warm water plankton are moving north and cold water plankton are moving out of the North Sea (reprinted from ICES, 2009 b).

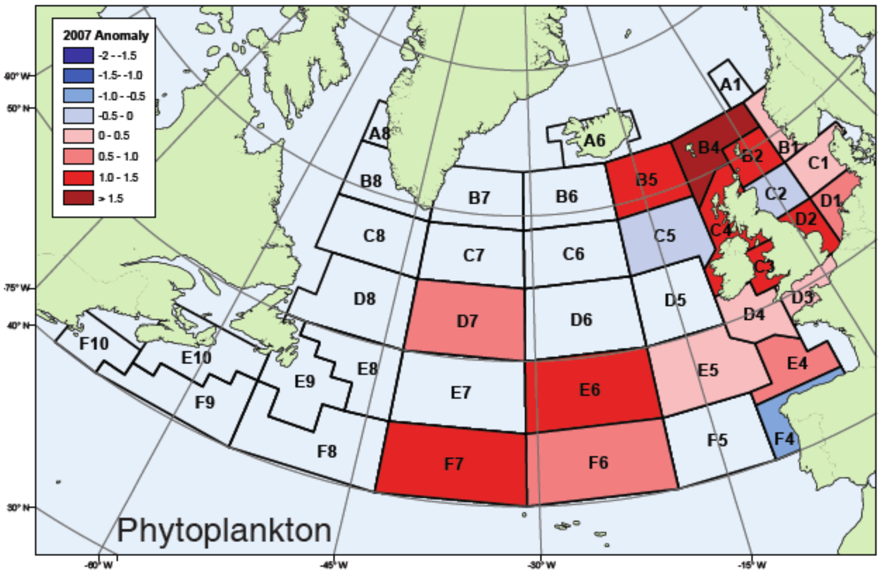


Figure 8.3.2.2. 2007 anomalies for the North Atlantic for phytoplankton biomass based on the long-term trends from 1958–2007. For phytoplankton biomass there has been a large increase since the late 1980s in most regional areas (particularly the North-East Atlantic). In 2007 the Phytoplankton Colour Index was generally above the base-line mean (1958–2007) in most regions, apart from some areas of the North Sea, central Atlantic and Iberian Peninsula. (Taken from: Edwards *et al*., 2009).

8.3.3 Conclusion and additional considerations

In summary, subject to changes in the selection of indicators (criteria), and reference levels on these (thresholds), there is no reason why plankton (species and communities) could not be included within the OSPAR QSR process. We consider this to be a desirable step if the assessment is to be seen to be a holistic ecosystem assessment. Inclusion of the plankton community would also require that any future operation of the OSPAR QSR process included interactions between components (making it a truly *integrated* ecosystem assessment). It is clear that changes in the plankton community and/or productivity could have major bottom-up effects on some of the other components, in particular the fish community, and these effects should be factored into the analysis. This could be achieved by including indicators that represent key ecosystem processes and functions (e.g., food chain dynamics indicators – see Section 3 of this Report).

It should be noted that the Management Group for the QSR ((MAQ)MAQ(2) 09/2/10 Add.3-E), also suggested that consideration should be given to other taxa, e.g., cephalopods and reptiles. It should be recognized that the number of ecosystem components chosen was narrowed considerably in the run up to the Utrecht meeting due to practical constraints. In any future QSR process the choice of components and the state indicators used to represent them should be based on an objective process as described in Section 3.4 (ToR a) of this Report.

8.4 Improving the method for working at different scales, such as the level of an OSPAR Region, the level of sub-regions such as the Irish Sea or the Channel or the level of an estuary or an MPA

8.4.1 Assessment at different spatial scales

Below we discuss some issues related to providing assessments at different spatial scales in relation to the OSPAR Matrix method. WGHAME (ICES, 2009b) provides an extensive overview, discussion and critique of these issues.

8.4.2 Information sources relative to scale considerations

Where available, locally collected data at the scale of the ecosystem of interest would be the most informative for assessing the ecological significance of a species or community property, setting objectives, selecting indicators, and setting management reference points. Frequently such data will not be available and in such cases information from other areas can be considered. The relevance of such data will vary with many factors, including similarity of spatial scales (e.g., Are coast-wide averages being applied to subsets of the region? Is a local study being extrapolated to a large area?), similarity of the ecological features being considered (e.g., What are the justifications for assuming that species and community properties played similar roles in the structure and function of the ecosystem where the information was collected and the one being evaluated now?), and even simple proximity (e.g., Are data being "borrowed" from adjacent areas or distant systems?). In all these cases, because of data limitations, uncertainty in the science advice will be higher, and this needs to be reflected through greater risk aversion in all management decision-making.

8.4.3 Some recommendations for assessment at different spatial scales using the Matrix approach

The Matrix approach (Robinson *et al*., 2008) was originally designed to assess the key pressures and impacts for a given habitat or species group. It has subsequently been adapted (Robinson *et al*., 2009) to a more general format for assessing status and impacts of pressure for Quality Status Reports within OSPAR. Below are some comments on assessments using the Matrix approach at different spatial scales:

- The methodology used in the OSPAR assessment (Robinson *et al*., 2009) was designed to be applied at any spatial scale, provided the thresholds (reference levels) set for each ecosystem component scaled with space. For example for any indicators of habitat components, reference levels set relative to a percentage decline will scale with space. As spatial scale decreases, however, the relevance to ecosystem components will change. For example, priorities set using the methodology within a very local area (e.g., an MPA) or estuary may not be applicable for species with high mobility.
- Some of the issues arising from the Utrecht workshop concerned the very large and ecologically diverse areas chosen for the evaluation. The key for this type of assessment would be to choose ecosystem components, and the scale of the ecoregions that are appropriate to management objectives. The impression from the Utrecht workshop was that the regions, and possibly the components had been determined from a socio-political perspective rather than matching components and regions to any underlying management or scientific rationale. This may, to some extent have undermined the credibility of the process. Essentially, the assessment methodology is designed to highlight key impacts at the scale that it is applied. Thus, if it is

applied across a very large region for a very broad habitat type, it will only identify pressures that are causing impacts at that scale.

- For the largest spatial scale, the Range and Extent descriptors will not be very useful for some of the major habitat types if habitat is considered in a very broad manner such as the sand habitat in the North Sea, which is not likely to change in range or extent. If, however, a finer habitat classification is applied such as the sandy mud or muddy sand areas (i.e., more resolved EUNIS levels), these descriptors may be more ecologically meaningful. For example, bottom trawling can modify or homogenize substrate. However, at those spatial scales the practicalities of mapping habitat distributions on scales of space and time that are fine enough to allow detection of change remain an impediment to using Range and Extent of habitats in all but local assessments.
- Another element to consider is the scale at which information is available for the particular species or habitat. For instance fish stock status information is detailed and comprehensive at the scale of say the North Sea, or indeed the Irish Sea, but is not particularly useful at the level of an MPA. To illustrate, we believe we can say how many cod are in the North Sea, but not in an estuary in that area. Conversely, habitat data may be much more detailed for an MPA than it is for the whole North Sea. Thus certain types of data might inherently be relevant for assessments at large scales but less so for local scales; and other types of data relevant at local scales much more than at larger scales.
- Several of the ICES-JRC Task Group reports and the Management Group Report also consider the issue of scale relative to individual Descriptors in implementation of the MSFD. Those reports should also be consulted when considering how to deal with scale issues in specific applications.
- A general concern with the application of the methodology was the lack of data to support it, and the data availability could therefore seriously affect the outcome of the analysis. It will therefore be valuable to perform a sensitivity analysis of how the results of the analysis depend in the input data, for example based on a boot strapping analysis. This is true in general, but sometimes may have to be considered as well when changing the scale of an assessment.

To conclude, given the points discussed above the Robinson *et al*. (2009) methodology is applicable at all the spatial scales mentioned in the request.

8.5 References

Awebro, K., Bager, M., Holm, P., Lajus, J., MacKenzie, B, and Ojaveer, H. 2002. Baltic Sea fisheries in previous centuries: Development of catch data series and preliminary interpretations of causes of fluctuations. ICES CM 2002/L:02.

Carlin, D. and Rogers, S. 2002. A Procedure to Assess the Effects of Dredging on Commercial Fisheries. Final Report of CSG Contract AO253. 84pp.

Desse, J. and Desse-Berset, N. 1993. Pèche et surpèche en Mèditerranèe: le temoinage des os. In Exploration des Animaux Sauvages à Travers le Temps, pp. 327–339. Ed. by J. DESSE and F. AUDOIN-ROUZEAU. Editions APDCA, Juan-les-Pins.

Edwards, M., Beaugrand, G., John, A.W.G., Johns, D.G., Licandro, P., McQuatters-Gollop, A. and Reid, P.C. 2009. Ecological Status Report: results from the CPR survey 2007/2008. SAHFOS Technical Report, 6: 1-12. Plymouth, U.K. ISSN 1744–0750.

Fletcher, W.J. 2005. The application of qualitative risk assessment methodology to prioritize issues for fisheries management. ICES Journal of Marine Science 62, 1576–1587.

Fosså, J.H., Mortensen, P.B. and Furevik, D.M. 2002. The deep-water coral *Lophelia pertusa* in Norwegian waters: distribution and fishery impacts. Hydrobiologia 471: 1–12.

Halpern, B.S., Selkoe, K.A. Micheli, F. and Kappel, C.V. 2007. Evaluating and ranking the vulnerability of global marine ecosystems to anthropogenic threats. Conservation Biology 21(5), 1301–1315.

ICES. 2009a. Report of the Working Group on the Ecosystem Effects of Fishing Activities (WGECO), 15–21 April 2009, Copenhagen, Denmark. ICES CM 2009/ACOM:20. 190 pp.

ICES. 2009b. Report of the Working Group on Holistic Assessments of Regional Marine Ecosystems (WGHAME), 12–16 October 2009, ICES Headquarters, Copenhagen. ICES CM 2009/RMC:13. 76 pp.

JNCC. 2007. Second Report by the UK under Article 17 on the implementation of the Habitats Directive from January 2001 to December 2006. Peterborough: JNCC: 39pp. Available from: www.jncc.gov.uk/article17.

Jennings, S., and Dulvy, N. K. 2005. Reference points and reference directions for sizebased indicators of community structure. ICES Journal of Marine Science, 62: 397–404.

Kenny, A. J., Skjoldal, H.R., Engelhard, G.H., Kershaw, P., and Reid, J.B. 2009. An integrated approach for assessing the relative significance of human pressures and environmental forcing on the status of Large Marine Ecosystems. Progress in Oceanography. 81, 132–148.

OSPAR. 2003. Criteria for the Identification of Species and Habitats in need of Protection and their Method of application (the Texel-Faial Criteria). OSPAR Convention Meeting Document 2003-13, Bremen, 23–27 June 2003; 13 pp.

Poulsen, B., Holm, P., and MacKenzie, B. R. 2007. A long-term (1667–1860) perspective on impacts of fishing and environmental variability on fisheries for herring, eel, and whitefish in the Limfjord, Denmark. Fisheries Research (Amsterdam) 87(2-3): 181–195.

Poulsen, B. 2010. The variability of fisheries and fish populations prior to industrialized fishing: An appraisal of the historical evidence. Journal of Marine Systems 79(3–4): 327–332.

Rice, J., Arvanitidis, C., Borja, A., Frid, C., Hiddink, J.G., Krause, J., Lorance, P., Ragnarsson, S.Á., Sköld, M., and Trabucco, B. 2010. Marine strategy framework directive - Task Group 6 report Seafloor integrity. Office for Official Publications of the European Communities, Luxembourg.

Rice, J. C. and E. Legace. 2007. When control rules collide: a comparison of fisheries management reference points and IUCN criteria for assessing risk of extinction. ICES Journal of Marine Science 64(4): 718–722.

Robinson, L. A. and Frid, C. L. J. 2008. Historical marine ecology: Examining the role of fisheries in changes in North Sea benthos. Ambio, 37: 362–371.

Robinson, L.A., Rogers, S., and Frid, C.L.J. 2008. A marine assessment and monitoring framework for application by UKMMAS and OSPAR-Assessment of Pressures Contract No:133 F90-01-1075 for the Joint Nature Conservation Committee). University of Liverpool, Liverpool and Centre for the Environment, Fisheries and Aquaculture Science, Lowestoft.108pp.

Robinson, L.A., Karman, C., Rogers, S., and Frid, C.L.J. 2009. Methodology for assessing the status of species and habitats at the OSPAR Region scale for the OSPAR Quality Status Report 2010. (Contract No: C-08-0007-0085 for the Joint Nature Conservation Committee). University of Liverpool, Liverpool and Centre for the Environment, Fisheries and Aquaculture Science, Lowestoft. 33pp.

Shin, Y-J., and Shannon, L. J. 2010. Using indicators for evaluating, comparing and communicating the ecological status of exploited marine ecosystems. 1. The IndiSeas project. – ICES Journal of Marine Science, 67: 000–000.

Smith A.D.M., Fulton E.A., Hobday A., Smith D., and Shoulder P. 2007. Scientific tools to support the practical implementation of ecosystem-based fisheries management. ICES J. Mar. Sci. 64: 633–639.

Tyler-Walters, H, Rogers, S.I., Marshall, C.E. and Hiscock, K. 2007. A method to assess the sensitivity of sedimentary communities to fishing activities. Aquatic Conservation: Marine and Freshwater Ecosystems. 19, 285–300

9 ToR g) Environmental interactions of wave and tidal energy generation devices (Marine wet renewables) (OSPAR request 2010/4)

g) Environmental interactions of wave and tidal energy generation devices (Marine wet renewables). OSPAR request 2010/4.

Provide advice on the extent, intensity and duration of direct and indirect effects and interactions of marine wet renewable energy production (wave, tidal stream and tidal barrage systems) with the marine environment and ecosystems of the OSPAR maritime area, and with pre existing users of these ecosystems, including:

i) actual and potential adverse effects on specific species, communities and habitats;

ii) actual and potential adverse effects on specific ecological processes;

iii) irreversibility or durability of these effects.

9.1 Introduction

While the term 'wet renewables' is commonly used to include offshore wind energy developments, the OSPAR request specifically defines the work to include only tidal barrages/fences, tidal stream and wave energy schemes. This reflects the greater level of knowledge and the more advanced stage of development of offshore wind energy schemes. These schemes however provide useful sources of information on environmental interactions that will be common across all (or most) schemes or that can be extrapolated to provide predictions for the effects of other types of scheme.

The various nations that border the OSPAR region are all committed to significant reductions in CO_2 emissions in the near term. Against this background energy demand continues to grow and restrictions on energy use are likely to be seen as economically and socially damaging. The challenge is therefore to move to a new low carbon economy where energy demands can be met while levels of CO_2 emitted are reduced.

For countries with significant areas of coastal waters the utilisation of offshore and coastal energy resources is attractive. The World Energy Council estimates that if less than 0.1% of the renewable energy within the oceans could be converted into electricity it would satisfy the present world demand for energy more than five times over (World Energy Council, 2007). The resources considered include the energy of the wind over the oceans, the waves and tides. While such headline grabbing figures are impressive the reality is that the technology does not exist to utilize most of the energy resource, not least because of issues associated with the spatial mis-match of the areas of demand with regions of highest resource.

The environmental issues associated with offshore wind farms have been well rehearsed (see Gill, 2005) and are not considered further here. As offshore wind energy developments have advanced attention has turned towards other sources of marine renewable energy that is less variable than that secured from wind.

For the purposes of this review we consider tidal energy associated with the change in water level in coastal bays, fjords or estuaries that might be harnessed by barrages or fences, tidal stream energy in tidal currents and the energy associated with waves.

The periodic change in water height associated with the tides has been used since at least Roman times to power mills for grinding grain. Modern tidal power schemes have sought to trap the high tide in an impoundment and then to run the water out through turbines in the barrage as the tide falls. This method is analogous to a hydroelectric facility. Electric generation from tidal height changes occurs commercially at the La Rance facility in France, operational since 1966, and the Annapolis Royal Power Station on the Bay of Fundy, Canada, operational since 1984. The 'oil crisis' of the 1970s stimulated interest in such schemes and in the UK a large research programme was commissioned to look at the engineering and environmental issues associated with a tidal barrage across the Severn Estuary. This culminated in a public enquiry lasting several years and the publication of the 'Bondi Report' in 1981 (Bondi, 1981).

Tidal barrages are essentially dams creating an impoundment. Tidal fences consist of a causeway across a bay or inlet and in place of a solid barrage the causeway is pierced by a series of turbines, that like the turnstiles at a major stadium, continually turn as the tide flood in and ebbs out of the basin. Tidal fences restrict the tidal regime but do not produce the large scale alterations that a barrage causes.

Tidal Stream farms use energy collection devices, tidal turbines or oscillating hydrofoils, mounted in regions of high flow to extract energy from the tidal currents. Wave energy collectors are usually surface mounded devices that capture the kinetic energy in the waves.

To date tidal fences, tidal stream farms and wave energy capture devices have only been deployed on an experimental scale and so prediction of their impacts is based on very limited empirical data.

9.2 Direct effects

9.2.1 Habitat change

9.2.1.1 Tidal barrage/fence

Tidal barrages work like hydroelectric dams except they need to allow water to flow in both directions. The sluice gates are opened to allow the tide to flood into a basin (estuary, fjord or bay), at high tide the sluices in the barrage are closed and the tide outside falls. Once a sufficient height difference has occurred the turbines are opened and the contained water flows out through the turbines. This continues until the tide turns and the differential head is eroded. The sluices are then opened to allow the basin to refill. This operation method, know as ebb generation, generates the most power. It is also possible to generate on the flood tide by refilling the basin through the turbines, while this generates power for more of the tidal cycle it generates less power in total as there is less of a differential head.

Building the barrage across the bay/estuary will destroy the former habitat in the development footprint. Construction and decommissioning will also probably result in impacts to adjacent intertidal areas used for construction of caissons or as staging areas. The presence of a barrage also influences habitats upstream and downstream of the facility. Upstream under ebb-only generation the upper intertidal remains submerged for a longer period, there is then a steady fall in tide level until the tide start rising again (Figure 9.2.1.1.1). The former lower shore remains submerged. These changes will shift the balance between marine intertidal species with upper shore specialists potentially being squeezed out. The retention of water also significantly alters the exposure of tidal flats to feeding birds although the resource in the tidal flats when they are exposed may increase in quantity and quality. The availability of alternative feeding/roosting sites is therefore often critical.

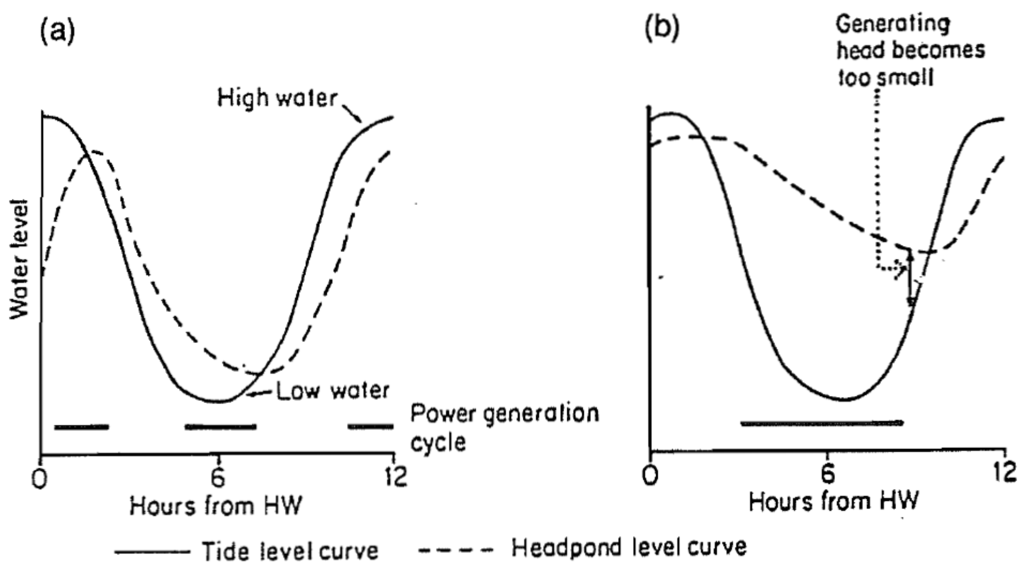


Figure 9.2.1.1.1. The normal tidal curve and the modified tidal curve in the headpond above a tidal barrage in an estuary for (a) dual cycle generating and (b) ebb only generation. (From Gray, 1992).

Downstream of the barrage tidal range is often reduced close to the barrage but enhanced in other parts of the basin (Wolf *et al*., 2009). The outflow will delay the falling tide from around mid-tide downward, such that the tide falls as normal, or more rapidly, from HW until the turbines open at mid-tide after which the rate of fall declines or is halted. This has potential negative implications for birds, although this effect occurs as the flats above the barrage become exposed.

Energy generation on the flood and ebb, dual mode, reduces considerably the changes in exposure of the intertidal area and so reduces potential impacts on the bird community (Figures 9.2.1.1.1 and 9.2.1.1.2.).

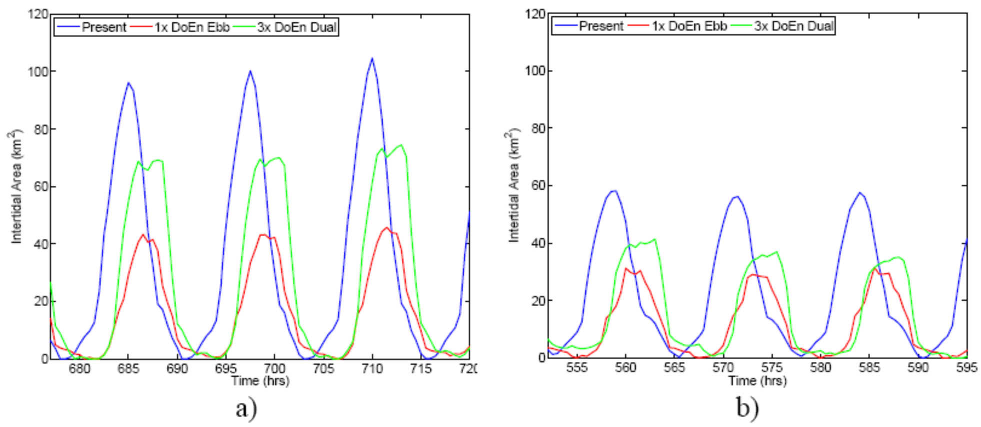


Figure 9.2.1.1.2. Changes in the area of intertidal flats exposed in the Severn Estuary on (a) Spring and (b) Neap tides under no barrage (Blue line), an Ebb-only generation scheme (red line) and a Dual mode barrage scheme (green line) (From Wolf *et al*., 2009).

The implications for tidally feeding fish are the opposite to those of the birds with greater periods for foraging available due to the retention/raising of water levels.

The economics of a barrage or fence scheme scale with the volume of the tidal prism and hence the most favoured schemes tend to involve large estuaries or bays. For example one options proposed for the Severn barrage in the UK would see 520 km^2 of the estuary impounded, compare this with the 17 km^2 at La Rance and 6 km^2 at Annapolis Royal. Given the very large environmental concerns with Severn development the smaller Mersey barrage may be the first in the UK to get regulatory approval. The Mersey scheme would involve and impoundment of 61 km^2 but even this would be sufficient to generate changes in the tidal range a t locations all around the Irish Sea (Wolf *et al*., 2009).

Changed spatial flow patterns will result in altered patterns of sediment deposition and movement. This will have impacts on benthic communities. The outflow will be constrained to a number of sites, where the turbines are, and in these areas sediments will be scoured and coarsened while upstream of the barrage the reduced flows and periods of no flow will lead to increased siltation and potentially an increasing quantity of fine material in the deposits.

Changes in the nature of the habitats will alter their suitability as nursery or spawning areas for fish. While some species may benefit from larger areas of appropriate conditions this still represents a deviation from the normal, pre-impact system.

The tidal fences are not expected to alter the timing or amplitude of the tides. During the construction phase it is inevitable that the natural benthic habitat will be destroyed and eliminated, but the structures themselves will create artificial habitat for benthic organisms.

9.2.1.2 Tidal stream farm

Energy generation using the tidal stream uses turbines or other devices placed in the water column to extract energy. The installation and operation of individual or multiple tidal stream devices, as with other forms of wet renewable energy systems, directly affects benthic habitats by altering water flows, wave structures, or substrate composition. Physical impact from small-scale tidal stream generation pilot projects have been found to be reversible on decommissioning, especially as the areas most suitable for tidal power generation are located where high current flow causes natural disturbance to the sediments. However, the cumulative effects of multiple turbines also need to be considered with respect to far field impacts.

During the construction phase of tidal stream farms the impacts on habitats will be similar to those experienced in the construction of other wet renewable installations. Bottom disturbances will result from the temporary anchoring of construction vessels; digging and refilling the trenches for power cables; and installation of permanent anchors, pilings, or other mooring devices. Fish and other mobile organisms will be displaced and sessile organisms smothered in the limited areas affected by these activities. Species with benthic-associated spawning or whose offspring settle into and inhabit benthic habitats are likely to be most vulnerable to disruption during installation.

Temporary increases in suspended sediments and sedimentation down stream from the construction areas can also be expected. When construction is completed, disturbed areas are likely to be re-colonized by these same organisms, assuming that the substrate and habitats are restored to a similar state. For example, Lewis *et al*. (2003) found that numbers of clams and burrowing polychaetes fully recovered within one year after construction of an estuarine pipeline, although fewer wading birds returned to forage on these invertebrates during the same time period.

Installation will alter benthic habitats over the longer term if trenches containing electrical cables are backfilled with sediments of different size or composition than the previous substrate. The use of large particles as a cover may be required to reduce the likelihood of cables becoming exposed and emitting electromagnetic fields into the water column. Permanent structures on the bottom (ranging in size from anchoring systems to seabed-mounted generators or turbine rotors) will smother existing habitats. These new structures would replace natural hard substrates or, in the case of previously sandy areas, add to the amount of hard bottom habitat available to benthic algae, invertebrates, and fish. This could attract a community of rocky reef fish and invertebrate species (including biofouling organisms) that would not normally exist at that site. It has been speculated that depending on the location, the newly created habitat could increase biodiversity or have negative effects by enabling introduced (exotic) benthic species to spread. Marine fouling communities developed on monopiles for instance in offshore wind power plants have been found to be significantly different from the benthic communities on adjacent hard substrates (Wilhelmsson *et al*., 2006; Wilhelmsson and Malm, 2008).

When operational, regardless of design and size, all tidal stream farms will include a large anchoring system made of concrete or metal, mooring cables, and electrical cables that lead from the offshore facility to the shoreline. Electrical cables may simply be laid on the bottom, or more likely anchored or buried to prevent movement. Movements of mooring or electrical transmission cables along the bottom (sweeping) have been shown to be a continual source of habitat disruption during operation. The strumming action of cables has been shown to cause incisions in rocky outcrops, but effects on seafloor organisms have generally be shown to be minor (Kogan *et al*., 2006). Large bottom structures will alter water flow and may result in localized scour and/or deposition. Because these new structures will affect bottom habitats, consequential changes to the benthic community composition and species interactions may be expected (Lohse *et al*., 2008).

Mobile bedforms resulting from the effects of new installations could modify the benthic habitat nearby, though the extent of these modifications depends on the character of the bottom in question. Tidal stream farms will likely be located in dynamic areas of exposed bedrock, which could reduce downstream drifting of sediment.

At this time, there are insufficient data to state definitively how fish and fish habitat will be impacted by the operation of tidal stream power projects. No published data on the interactions between turbines and fish in the marine environment could be found except for some information from the Roosevelt Island tidal energy project (Anon, 2008). It is generally felt that fish avoidance of tidal stream installations should be possible, and proponents of the technology suggest that the rotation speed of the turbines will be slow enough to be avoidable by fish. However, it is not clear whether some species utilize high currents to passively carry them, making avoidance of the devices more difficult. The study in the Roosevelt Island showed that densities observed in and around the turbines was generally low (range of 16–1400 fish per day seen); the fish were predominantly small but still swam faster than the turbines rotated; and fish movement tended to be restricted to the direction of the tide and during slack water when the turbines were non-operational (Anon, 2008).

Tidal stream farms operate in a very different manner to hydroelectric and tidal barrage systems. In the latter a high speed turbine is mounted in a tunnel through which water flows at high speed and considerable pressure. Thus entrained organisms have little or no chance of avoiding passing thorough the turbine. In tidal stream farms the devices may not involve rotary turbines at all. Some devices for example rely on the see-saw oscillation of a beam with hydrofoils at each end. When rotary turbines are used they are mounted in the open flow field and so the rate of revolution is much lower and organisms have plenty of opportunity to avoid direct contact.

There remain large information gaps concerning the collision risk of marine mammals with static structures such as tidal stream farms. The literature reviewed suggests that the probability of cetaceans failing to detect and avoid a large static structure is considered to be extremely low, particularly for species that echo-locate and are agile and quick moving. The environmental report produced as part of the Roosevelt Island tidal energy project reported that the largest potential risk to marine mammals would be if a species moving through the area was directly struck by a turbine blade, potentially causing injury or mortality. Boat propeller strikes have been reported to cause mortality to mammals and turtles but the blades on the Roosevelt Island operation rotate at much slower speeds than a typically boat propeller so it is doubtful this is an actual risk. The exact placement of tidal farms for species that frequent particular areas, either through site fidelity or seasonally, should be considered in mitigation. Feeding and breeding sites in particular for marine mammal species should be avoided when tidal farm sites are selected. This is logical risk management strategy in the face of uncertainty even though there are no documented cases of any negative impact on marine mammals.

The impacts of tidal stream farms on seabirds are also reported to be small. Risk of collision is expected to be minimal as for many species of sea birds, including gulls, terns, kittiwakes, fulmars and skuas, their normal depth range would not allow them to encounter operating turbines. For some deep diving species, e.g. auks, shags, there is the chance of an encounter as these species regularly dive to depths of 45–65m. The critical issue is the relative swimming speed of the bird, and the ability to sense and respond to the turbine. The slow turbine speeds relative to the agility of diving bird species is would make the risk of mortality very low (Awatea, 2008). A typical swimming speed for these species is of the order of 1.5 ms^{-1}. For comparison, the tip of a 2 m diameter turbine turning at 15 rpm would be moving faster than this and so potentially be difficult for a bird to avoid. In the Roosevelt Island tidal energy project, based on quite extensive observations both pre and post deployment of the structures, no negative impacts were observed (Anon, 2008). The possible interactions are further complicated by the possibility that diving birds may respond to the moving blades as potential prey and be attracted to their vicinity. Further work is needed to elucidate the scale of this phenomenon and to develop mitigation measures, i.e., painting the blades.

9.2.1.3 Wave energy farm

Wave energy farms show a wide variety of systems, at several stages of development, competing against each other, without it being clear which types will be the final winners (Falcão, 2010). Some offshore wave energy farms are expected to contribute to an increase in submerged constructions on the seabed, including a possible impact on the surrounding soft-bottom habitats. As both pilot and commercial wave energy converting applications are limited, so are studies on habitat change. One Swedish study details the effects over a five-year study period after the construction of wave energy constructions (Langhamer, 2010). The author concludes that the wave energy converters had only minor direct effects on the benthic community (macrofaunal biomass, densities, species richness and biodiversity) in relation to the natural high variations. However, concrete foundations might displace activities such as trawling, which may contribute to stabilizing the seabed. This possibly induces positive long-term effects on soft-bottom macrofauna such as an increase in their biomass and trophic compositions (Langhamer, 2010). Langhamer and Wilhelmsson (2009) examined the function of wave energy foundations as artificial reefs. They found that fish numbers were not influenced by increased habitat complexity (holes), but a significantly positive effect on quantities of edible crab (*Cancer pagurus*) was found. Densities of spiny starfish (*Marthasterias glacialis*) were negatively affected by the presence of holes, potentially due to increased predator abundance. The authors suggest a species-specific response to enhanced habitat complexity. Langhamer *et al*. (2009) demonstrated that foundations serve as colonisation platform with a higher degree of coverage on vertical surfaces. Buoys were dominated by the blue mussel *Mytilus edulis*.

Larval distribution and sediment transport can both change. Additionally, the fouling community growth on buoys, anchors, and lines may adversely affect the benthic environment if deposited into accumulations on the seafloor.

Some authors have speculated that changes in surface productivity linked to a reduced mixing may alter the food supply to benthic populations (Pelc and Fujita, 2002). The extent to which wave energy farms will reduce water column mixing or the amplitude of waves impinging on to coastal habitats is unknown.

Regarding the pelagic habitat, buoys have a minimal impact on phytoplankton, but positive effects on forage species, which consequently cause an attraction of large predators. On the other hand, lines on structures can cause the entanglement of marine mammals, turtles, larger fish and seabirds, but they also can produce an increase of settlement of meroplankton (Boehlert *et al*., 2007; DFO, 2009).

The dampening of waves may reduce erosion on the shoreline; whether this effect is considered positive will depend on the societal and environmental value of the coastline. Dampening may cause ecological changes but sheltering due to wave devices will have a negligible effect on the largest waves, so that the ecological role of very large waves as a disturbance that maintains high biodiversity will be unencumbered (Pelc and Fujita, 2002).

9.2.2 Water column processes and hydrography

9.2.2.1 Tidal barrage/fence

Downstream of the barrage during outflow and immediately upstream on inflow, the constraining of the flow will lead to turbulent flows that will increase mixing. Upstream for much of the tidal cycle the water in the basin will be fairly static and this could lead to stratification in summer, and changes in the phytoplankton dynamics.

In the Severn Estuary, for example, the strong tidal flows lead to highly turbid conditions and hence low primary productivity. Underwood, 2010 suggested that following construction of a barrage the increased water clarity upstream could lead to increased phytoplankton derived primary production. However, this is thought to be less than the loss of primary production from microbial primary producers in the sediments which is decreased following the impounding of water and the reduction in emergent area of tidal flats.

Studies of the impact of passage through turbines on marine plankton are currently lacking. Reported mortality of freshwater zooplankton following entrainment in hydroelectric turbines can be high (Jenner *et al*., 1998). However, in many estuaries where tidal ranges are large plankton populations are low and derived from individuals advected in. This suggests that even if mortality of entrained individuals is high this is not likely to be significant at the population and community level.

Levels of direct mortality of fish passing through turbines can be high and the disorientation caused may lead to lowered ability to avoid predation in the period after passage. However there is considerable experience of engineering sluices, cooling water intakes and turbines to reduce fish entrainment (Coutant and Whitney, 2000) and such mitigation measures should be seen as a critical part of any system design.

Energy extraction may affect turbulent mixing, and change patterns of sediment distribution. Tidal fences in high energy coastal areas may encounter currents moving at 5 to 8 knots (9 to 15 km per hour) producing intense mixing processes continuously in the water column. At lesser velocities some degree of water column stratification can be expected (Gray, 1992). This may also bring increased water clarity through reduced sedimentation.

9.2.2.2 Tidal stream farm

Tidal energy power generation devices will increase turbulence in the water column, which in turn will alter mixing properties, sediment transport and, potentially, wave properties. In both the near field and far field, extraction of kinetic energy from tides will decrease tidal amplitude, current velocities, and water exchange in a region in proportion to the number of units installed, potentially altering hydrography and sediment transport. The effect on transport and deposition of sediment may also influence organisms living on or in the bottom sediments, and plants and animals in the water column. Moving rotors and foils have been shown to increase mixing in systems where salinity or temperature gradients are well defined.

Changes in water velocity and turbulence will vary greatly, depending on distance from the structure. For small numbers of units, the changes are expected to dissipate quickly with distance and are expected to be only localized; however, for larger commercial arrays, the cumulative effects will extend to a greater area although it is still not know whether these would have significant effects on the ecosystem.

Tidal energy turbines may also modify wave heights by extracting energy from the underlying current. The effects of structural drag on currents are not expected to be significant (MMS, 2007), but few measurements of the effects of tidal/current energy devices on water velocities have been reported. Tidal velocity measurements were made near a single, 150 kilowatt (kW) Stingray demonstrator in Yell Sound in the Shetland Islands (The Engineering Business Ltd., 2005). Acoustic Doppler Current Profilers were installed near the oscillating hydroplane (which travels up and down in the water column in response to lift and drag forces) as well as upstream and downstream of the device. The data suggested that tidal currents of 1.5 to 2.0 m/s were slowed by about 0.5 m/s downstream from the Stingray. In practice, multiple units will be spaced far enough apart to prevent a drop in performance (turbine output) that may result from extraction of kinetic energy and localized water velocity reductions.

Changes in water velocities and sediment transport, erosion, and deposition caused by the presence of new structures will alter benthic habitats, at least on a local scale. Craig *et al*. (2008) reports that deposition of sand may impact seagrass beds by increasing mortality and decreasing the growth rate of plant shoots. Conversely, deposition of organic matter in the wakes of tidal farms could encourage the growth of benthic invertebrate communities that are adapted to that substrate (Widdows and Brinsley, 2002). While the new habitats created by such structures may enhance the abundance and diversity of invertebrates, predation by fish attracted to artificial structures can greatly reduce the numbers of benthic organisms (Davis *et al*., 1982; Langlois *et al*., 2005).

9.2.2.3 Wave energy farm

Wave power plants act as wave breakers, calming the sea, and the result may be to slow the mixing of the upper layers of the sea, which could cause an adverse impact on the marine life and fisheries (Pelc and Fujita, 2002). The energy devices remove energy from the wave train, affecting the tidal range, sediment deposition and ecosystem productivity. Similarly, erosion patterns along long stretches of coastline could be changed, being the effect beneficial or detrimental depending on the specific coastline (Pelc and Fujita, 2002). They may also modify some other local sediment transport patterns (including re-suspension and deposition) by localized hydrodynamic changes due to presence of physical structures and from energy extraction. And depending on the location, scale, technological characteristics and dynamical processes, all these effects can be extended along the environment. Substrate disturbance during deployment, decommissioning and maintenance processes, for example, can lead to increased suspended sediments and turbidity, especially in areas with finer substrates such as sand or silt. Sediment re-suspension may directly cause deleterious health effects or mortality to fish, and increased turbidity could hinder the prey detection ability of species that rely on visual cues (DFO, 2009). All these processes could alter the way the ocean interacts with the atmosphere locally but given the scale of the ocean they are unlikely to be of ecological significance for system functioning (Pelc and Fujita, 2002).

Maintenance involves the use of service boats regularly, sometimes small boats, but some other times, big boats able to transport devices to a port if needed. It also carries some risk of shipping accident, which consequences are well known these days. Decommissioning could cause more bottom disturbance than deployment. But the possibility of creating an artificial reef could be considered, using some of the structures, like anchors (DFO, 2009).

9.2.3 Exclusion zones

9.2.3.1 Tidal barrage/fence

The presence of the barrage or fence will result in, probably a 0.5 nautical mile exclusion on either side for fishing vessels, vessels anchoring, etc. On most large barrage proposals the passage of shipping through the barrage is maintained by the provisions of appropriate lock systems with associated breakwaters and channels. Thus the effect of exclusions zones is minimal for most users.

Exclusion zones will be required during both construction and operation phases. They would likely be larger during the construction period and reduced once the system was operational.

9.2.3.2 Tidal stream farm

It is likely that tidal stream farms will have exclusion zones within and around them to provide a safety barrier from other activities, such as fishing and navigation, similar to those found at other marine energy structures. Exclusion zones are likely to be marked by cardinal buoys and navigation lights, noted on shipping charts in future and advised through Notices to Mariners. Whilst other human activities are likely to be excluded in the area of marine energy converters arrays, the exclusion zones may create *de facto* marine reserves, in which marine life can flourish. The nature of the changes associated with marine protected areas are not simple to predict but there is a considerable body of data showing the effects of such schemes (Balmford *et al.*, 2004; Murawski, 2005; Murawski *et al.*, 2005; Kaiser, 2005; Rice, 2005).

Industrial sectors such as fishing and shipping are likely to have concerns regarding both spatial exclusions around tidal stream farms, as they do with other renewable energy projects. Exclusion zones may also impact indirectly as they may lead to displacement of such activities to other areas. Marine energy projects will add to the cumulative impact of closures for other reasons.

9.2.3.3 Wave energy farm

Commercially operated wave energy farms are limited (e.g., Portugal and Scotland). Therefore, one can only speculate about possible configurations (e.g., Falcão, 2010). Length and width vary by number and type of Wave Energy Converters (WECs). The Pelamis-type for instance needs a total of 150 m, whereas 15 m is sufficient for the AWS-type. However, WECs are usually deployed in multiples and the footprint will therefore vary with the actual configuration. These differ and accordingly the exclusion of other users will. Effects on the ecosystem might change due to displacement of fishing effort, changes in migration paths of marine mammals, altered shipping, etc. (Boehlert *et al*., 2007). However, from the nature conservation perspective, the exclusion is likely to have some effects similar to those associated with the developing networks of marine protected areas (Langhamer *et al*., 2010). Construction of wave energy farms temporarily affects harbour seals when rock and isles used for resting are within a short distance of the wave energy farm (Langhamer *et al*., 2010). The importance of the altered effects depends on the extent of the displacement.

9.2.4 Noise

9.2.4.1 Tidal barrage/fence

Operational noise is unlikely to be ecologically significant. However barrages are major civil engineering structures and construction (and decommissioning) activities will include considerable noise generating activities at levels potentially damaging to marine life.

During construction noise and vibrations would affect different species in different ways (US Department of Energy, 2009; DFO, 2009). Pile driving would likely affect schooling fish or any species with a swim bladder. Effects on other species would be less certain. Effects could be direct, by damaging sensory or sensitive tissues, or indirect, by changing behaviours. Possible effects on marine mammals could include construction effects of noise, vibration and lights; noise and vibration during operation affecting species that use sonar to pursue prey or affecting communication between animals; direct collision or contact; and indirect effects on the distribution and abundance of prey species. Migratory shorebirds depend on benthic intertidal invertebrates, the abundance and distribution of which might be altered by tidal development through sediment changes. During the operations phase noise and vibrations could continue to affect some species. It is important when assessing noise effects that the cumulative effects of the entire system be evaluated and not just the levels produced by individual modules (US Department of Energy, 2009).

Activities likely to produced noise at levels of concern include pile-driving, explosive or seismic work. Even within the construction/decommissioning phases these are intermittent, short duration activities but they have the potential to effect cetacean or pinniped activity in the region at the same time (Madsen P.T. *et al*., 2006). At offshore wind farms in Denmark, Henriksen *et al*. (2004) and Tougaard *et al*. (2003) both found effects on the behaviour and abundance of harbour porpoises during pile driving activities. Fewer animals exhibited foraging behaviour and there was a short-term reduction of echolocation activity. These effects were documented up to 15 km from the impact area. These effects were, however, short-lived once construction ceased (Carstensen *et al*., 2006). Studies suggest that high-level impulsive sounds have a greater effect on cetaceans than pinnipeds (McCauley and Cato, 2003; Gordon *et al*., 2004).

9.2.4.2 Tidal stream farm

There is very little information on the sound levels produced by the construction and operation of tidal stream farms. If installation involves pile driving, which most pilot projects have done, nearby noise levels are likely to exceed threshold values for the protection of fish and marine mammals. Operational noise from a small number of units may not exceed threshold levels, but the cumulative noise production from large numbers of units has the potential to mask the communication and echolocation sounds produced by aquatic organisms in the vicinity of the structures.

There are considerable information gaps regarding the effects of noise generated by tidal stream farms on cetaceans, pinnipeds, turtles, and fish. Sound levels from these devices have not been routinely measured, but it is likely that installation will create more noise than operation. Operational noise from generators, rotating equipment, and other moving parts may have comparable frequencies and magnitudes to those measured at offshore wind farms; however, the underwater noise created by a wind turbine is transmitted down through the pilings, whereas noises from tidal stream farms are likely to be greater because they are at least partially submerged. It is probable that noise may be less than the intermittent noises associated with shipping and many other anthropogenic sound sources (e.g., seismic exploration, explosions, commercial, naval sonar) but this is based mainly on conjecture rather than extensive physical measurement.

Resolution of the significance or otherwise of noise impacts will require information about the device's acoustic signature (e.g., sound pressure levels across the full range of frequencies) for both individual units and multiple-unit arrays, similar characterization of ambient noise in the vicinity of the farm, the hearing sensitivity of fish and marine mammals that inhabit the area, and information about the behavioural responses to anthropogenic noise (e.g., avoidance, attraction, changes in schooling behaviour or migration routes).

9.2.4.3 Wave energy farm

A large number of species of different taxa (cetaceans, pinnipeds, teleosts, crustaceans) use underwater sounds for interaction and echolocation (Misund and Aglen, 1992; Popper and Hastings, 2009; Langhamer *et al*., 2010). There have been very few (if any?) directed studies of the response of fish and marine mammals to noises and vibrations produced by operational WECs (DFO, 2009). DFO, 2009 reports existing modelling studies suggesting construction and operation noise levels can cause temporary, or in certain circumstances, permanent hearing loss in porpoises, seals and some fish and interfere with interactions between organisms (communication, finding prey, location of recruitment sites, etc.). Langhamer *et al*. (2010) remark that the production of noise by drilling and placing during construction, cable laying, as well as boat traffic can damage the acoustic system of species within 100 m from the source and cause mobile organisms to avoid these areas during that time. However, the authors also suggest that placing of gravity foundations on the seabed may have little effect on acoustic sensitive organisms. Behavioural reactions of marine mammals to noise due to construction and operation are highly variably since habituation cannot be ruled out as well as exposure to many other noise-sources (Langhamer *et al*., 2010). As for other effects, the type of WECs and scale of application determines the production of noise and subsequent effects (Boehlert *et al*., 2007). The constant low-intensity sounds from operating WECs have also been compared to light to normal density shipping and a conventional ferry or subway (Anon, 2008), implying that effects may also be of a comparable magnitude. Understanding of the long-term effects of noise is limited.

9.2.5 Electromagnetic fields (EMFs)

All types of cable will emit electromagnetism to the surrounding water. The electric current travelling through the cables will induce magnetic fields in the immediate vicinity, which can in turn induce a secondary electrical field when animals move through the magnetic fields (CMACS, 2003). Damage to the electrical transmission cable could cause an electrical fault or short, during which electrical current would leak to the water.

9.2.5.1 Tidal barrage/fence

Electricity generated by the existing barrage facilities is carried away by cables running on the top of the barrage and so has no marine environmental impact.

9.2.5.2 Offshore energy installations-tidal stream and wave energy farms

The environmental impacts of electromagnetic emissions from cables, switch gear and sub-stations is the same irrespective of the energy generating device and thus the lessons learnt from offshore wind power developments are applicable to developments harnessing tidal stream or wave energy.

The current state of knowledge about the EMF emitted by submarine power cables is too variable and inconclusive to make an informed assessment of the effects on aquatic organisms (CMACS, 2003). Following a thorough review of the literature related to EMF and extensive contacts with the electrical cable and offshore wind industries, Gill *et al*. (2005) concluded that there are significant gaps in knowledge regarding sources and effects of electrical and magnetic fields in the marine environment. They recommended developing information about likely electrical and magnetic field strengths associated with existing sources (e.g., telecommunications cables, power cables, electrical heating cables for oil and gas pipelines), as well as the generating units, offshore sub-stations and transformers, and submarine cables that are a part of renewable energy projects, including tidal stream farms. They cautioned that networks of cables in close proximity to each other (as would be found in commercial scale tidal energy projects where cables come together at substations) are likely to have overlapping, and potentially additive, EMFs. These combined EMFs would be more difficult to evaluate than those emitted from a single, electrical cable (CMACS, 2003).

It is well documented that several marine species use magnetic and electrical fields for navigation and locating prey. Electrical fields (E fields) are proportional to the voltage in a cable, and magnetic fields (B fields) are proportional to the current. All fish are sensitive to a greater or lesser extent to electric fields. Sharks and rays in particular may find their prey using the weak field emitted by fishes (Kalmijn, 1982) and may employ electromagnetic fields for navigation (Paulin, 1995). Electro-sensitive species may be either attracted or repelled by such fields, depending on their strength (Kalmijn, 1982; Gill, 2005). In a typical industry-standard cable, typically used in tidal stream farms, conducting 132 kV and an AC current of 350 A, the size of the B field would be of low magnitude: ca. 1.6 μT (micro-Tesla) and present only directly adjacent to the cable. It has been shown that such a field would fall to background levels (ca. 50 μT) within 20 m of the cable (CMACS, 2003). Some species of shark have been shown to respond to localized magnetic fields of 25–100 μT (Meyer *et al*., 2004). Marra (1989) showed that induced E fields of upto 91 μV were emitted from cables buried to

1 m in sediment. It is entirely possible that benthic-foraging elasmobranchs (especially rays) may detect and react to even weak E fields from AC cables. It is, however, equally likely that the range of influence of the field will be limited, and appropriate shielding measures will reduce the likelihood of deleterious effects. Cables carrying direct current (DC) from individual installations are likely to carry only 10–15 kV, which is unlikely to generate any electrical field more than a few cm from the cable, especially if a 3-phase carrier is used. However, high voltage DC cables may produce fields of up to 5 μT at up to 60 m (Westerberg and Begout-Anras, 2000). More recently, Westerberg and Lagenfelt (2008) found evidence that a 3-phase 130 kV cable (unburied) may be detected by migrating European eels *Anguilla anguilla* but did not disrupt their migration. Although some marine animals such as turtles may use the Earth's magnetic field for navigation (Lohmann and Johnsen, 2000), evidence for marine mammal utilization is equivocal (Hui, 1994). Such limited range fields are unlikely to be detected by pelagic species.

It has also been shown that tidal stream farms are unlikely to create magnetic fields strong enough to cause physical damage to marine organisms. For example, Bochert and Zettler (2004) summarized several studies of the potential injurious effects of magnetic fields on marine organisms. They subjected several marine benthic species (i.e., flounder, blue mussel, prawn, isopods and crabs) to static (DC-induced) magnetic fields of 3700 μT for several weeks and detected no differences in survival compared to controls. In addition, they exposed shrimp, isopods, echinoderms, polychaetes, and young flounder to a static, 2700 μT magnetic field in laboratory aquaria where the animals could move away from or toward the source of the field. At the end of the 24 hour test period, most of the test species showed a uniform distribution relative to the source, not significantly different from controls. Based on these limited studies, Bochert and Zettler (2004) concluded that they could not detect changes in marine benthic organisms' survival, behaviour, or a physiological response parameter (e.g., oxygen consumption) resulting from magnetic flux densities that might be encountered near a typical undersea electrical cable.

Lohmann *et al*. (2008) does report that given the important role of magnetic information in the movements of sea turtles (particularly loggerhead turtles), impacts of magnetic field disruption could range from minimal (i.e., temporary disorientation near a cable or structure) to significant (i.e., altered nesting patterns and corresponding demographic shifts resulting from large-scale magnetic field changes) and they suggest that this should be carefully considered when sites for tidal farms are authorised.

Studies of the European eel's positioning in an electro- magnetic field, conducted under controlled laboratory conditions, indicate that the eel can use magnetic navigation in a small scale to find its way from the stream through a complex coastal geography, as well as sense it's global position due to the strength and angle of the magnetic field (Hauge, 2010; Durif *et al*., *in preparation*).

The survival and reproductivity of several benthic organisms is not affected by long-term exposure to static magnetic fields (Bochert and Zettler, 2004), nor seem fish affected to any significant degree by sea cables and their magnetic and electric fields (Gill, 2005; Gill and Taylor, 2001; Öhman and Wilhelmsson, 2005).

Langhamer *et al*. (2009) remarks that with the use of a better cable technology the electromagnetic fields only affect the nearest surroundings as the background earth magnetic field usually becomes more prominent only a few decimetres from the cable. In combination with cables buried into the seabed, issues with electromagnetic fields might disappear.

9.2.6 Contaminants and anti-fouling

With regard to water quality, the loss of oil is the biggest impact identified. Subsurface electrical equipment will contain oil as an insulator and lubricant while some designs of wave and tidal stream energy collection devices use hydraulic systems that will contain oil. Modular design and appropriate valves should limit the volume of oil loss in the event of a structural failure or collision damage (Boehlert *et al*., 2007). Modern materials used in manufacturing and the regulations regarding placement in the marine environment will limit the risk of the devices introducing contamination into the sea (Boehlert *et al*., 2007; DFO, 2009).

One potential source of contamination is leaching from anti-fouling preparations. High speed moving surfaces are unlikely to require protection while large areas may not need to be protected and fouling communities will actually contribute to the biodiversity value of the development. Some areas will however need, for operational efficiency, require antifouling protection. Modern anti-fouling preparations tend to be low toxicity and biodegradable. Given that both wind and wave devices will be deployed in high energy environments this seems little likelihood that the ecologically significant effects will occur.

9.3 Indirect effects

9.3.1 Food chain

9.3.1.1 Tidal barrage/fence

The principle food chain effect of tidal barrages is the reduction in infaunal food to the bird population. The larger the scheme the more likely it is that that there will not be alternative feed sites nearby. In the UK probably northern Europe in general the quantity and quality of the food on the feeding grounds of over wintering waders is the parameter that determines survival to the next breeding season. Thus reduced feeding areas, increased foraging costs (extra flights between sub-optimal grounds) or lower food quality will directly impact on population size. The greater foraging time available to fish predators in the intertidal may also alter species composition of the fish assemblage by favouring species able to exploit this resource efficiently.

There is some evidence downstream of hydroelectric dams in some freshwater systems of the build up of detritus, derived from moribund and deceased plankton impacted by the passage through the turbines (see Jenner *et al*., 1998 and references therein) but in a dynamic estuary any such effect is likely to be widely dispersed and rapidly used by the detritivores.

9.3.1.2 Tidal stream farm

Principle indirect effects of tidal power turbines will relate to the consequences for biota of local physical impacts, and to changes in hydrographic conditions that may result from tidal energy extraction. Few studies have been undertaken which help to specify the magnitude or importance of such effects, beyond those generic indirect effects resulting from the placement of structures on the seabed.

9.3.1.3 Wave energy farm

Wave energy arrays provide a matrix of hard structures development, which will likely have ecological consequences from the fouling community up through the highest levels of trophic structure. Moreover, forage species are attracted by these devices, which is associated with an increase of presence of large predators and the corresponding changes in the food web.

Some marine species (cetaceans, pinnipeds, teleosts, crustaceans) are especially sensitive to acoustics (Popper and Hastings, 2009). Avoidance of areas by certain species or changes in foraging success due to interactions between anthropogenic noise with acoustic sensory apparatus could result in food chain effects (Boehlert *et al.*, 2007). The structural complexity that these devices give to the marine environment will alter the habitat and hence the trophic relationships by for example providing opportunities for ambush predators, shelter for prey, while the presence of organisms attached to or hiding between the structures that may serve to increase the range of potential prey items available (Langhamer *et al.*, 2010).

9.3.2 Reproduction and recruitment

9.3.2.1 Tidal barrage/fence

Construction of a barrage on or near a nursery or spawning area will clearly have an impact. These are site specific considerations. More generally by producing a barrier across the estuary/fjord the barrage will impact on migrations of anadromous and catadromous species including economically importance salmonids and eels and societally important species such as shad. Mitigation using salmon ladders is well developed and proven technology for hydroelectric dams.

Tidal fences will also restrict fish and marine mammal passage through physical blockage, although there is room for mitigation through engineering of the fence structure to allow spaces for fish to pass through between the caisson wall supporting the turbines and the rotors. Further, placement of the fence (in-parallel or in-series to water flow) can greatly influence impacts on species and habitats. There are some claims that if the rotors move slow enough the fish can move through without physical damage and study of a 20 kW prototype built in 1983 by Nova Energy, in the St. Lawrence Seaway, Canada reported no fish kills (Pelc and Fujita, 2002). Turbine velocities in the range of 25–50 rpm are expected to minimize fish kills from physical contact with the blades (Pelc and Fujita, 2002).

Marine mammals will be attracted to the devices in search of fish that are killed or disoriented and may be caught in the rotors as they attempt to hunt. Mitigation measures to prevent marine mammals have been proposed and range from physical barriers around the blades (which themselves may cause further environmental problems) to more sophisticated solutions such as sonar sensor systems that shut down the turbines when marine mammals are detected (Pelc and Fujita, 2002).

9.3.2.2 Tidal stream farm

Reproduction of organisms with short-lived, mobile early life history stages will only be affected if settlement is hindered by local changes in turbulence or tidal stream direction. The reproduction of species with longer lived egg and larvae stages are unlikely to be affected unless multiple devices are very closely packed.

9.3.2.3 Wave energy farm

Many fish species depend in part on currents to transport larvae, so wave energy devices that alter the currents between spawning grounds and feeding grounds could be harmful to fish populations (Boehlert *et al.*, 2007). On the contrary, and based on the fact that some of these structures become artificial reefs where biodiversity increases, food availability increases and feeding efficiency is also higher, which could cause an enhancement of the larval recruitment in the area (Sánchez-Jerez *et al.*, 2002). A complex substratum increases the spatial heterogeneity which can increase the species diversity of an area by providing more ecological niches, allowing more animals to recruit (Menge, 1976).

It has been hypothesised that noise might interfere with the ability of some fish species that locate their nursery areas by sound (Langhamer *et al*., 2010) although specific data were not presented. Breeding vocalizations are important for mate attraction in freshwater goby (Lugli *et al*., 1996), cod (Finstad and Nordeide, 2004) and haddock (Hawkins and Amorima, 2000). The successful settlement of coral reef fish depends on reef noise and can be affected by noise pollution (Simpson *et al*., 2008).

9.4 Principle areas of environmental risk and the scope for mitigation

Tidal barrages and to a slightly lesser extent tidal fences are extremely capital intensive and their coastal location means that they are subject to major planning and environmental regulatory approval requirements. This means that such developments are likely to be restricted to areas of high return (i.e., large tidal prisms) and where societal need is sufficient to over-ride planning and environmental concerns. The principle environmental effects of a barrage are the changed tidal regime and its impact on bird communities and habitat availability. The impacts on habitats are not easily mitigated; a certain degree of loss of the regional habitat pool is inevitable. The impacts on bird feeding habitat can be mitigated by the provision of new intertidal areas/lagoons which provide feeding grounds during the high water period landward of the barrage, the use of a dual cycle generation regime or the substitution of the barrage by a tidal fence. The latter options both give a lower energy yield. If the site was on a fish migration route (salmonids, eels, shad) appropriate provision would need to be provided by means of fish passes, etc.

The impact of tidal stream energy generation on the marine environment will be broadly equivalent to that of offshore wind farms in their construction, operation and spatial footprint. Although the site-specific impacts of turbine construction and operation will be limited, the placement of multiple turbines in offshore or coastal farms may exclude many other marine sectors. This potential exclusion will need to be carefully managed to avoid conflicts with other sea users, such as shipping lanes, marine aggregate extraction, etc., where site specific requirements for access may also exist. Protection of the locality from potentially harmful activities may increase habitat diversity and provide a significant contribution to site-based marine protection. Unlike wind farms, there may not be much visible infrastructure above the water surface, so hindrance to normal marine navigation will need to be carefully managed.

Although of potential concern, there is little scientific literature to suggest that operation of underwater tidal stream energy devices will cause elevated levels of mortality to pelagic organisms such as fish and marine mammals.

The effects of wave energy farms are poorly understood, making it hard to prioritize areas of environmental risk. The deployment of wave energy farms (WEF) will potentially lead to change in benthic and pelagic habitat characteristics in several ways. Selecting sites for WEFs causes the displacement of activities such as fishing, substantially changing the pressures on the seabed in the selected site as well as in the areas where the other activity is displaced to. Additionally, the structures associated with the WEF will change the habitat complexity. The pelagic habitat is changed by creating platforms for predators, e.g. seabirds, and by changing the hydrographical conditions. These changes in habitat and subsequently in species composition will lead to altered food web dynamics. The influence of WEFs on benthic and pelagic habitats is not precisely quantified and should therefore be carefully investigated. Deploying WEFs occasions a removal of wave energy and modification of the current flow.

Mitigation measures should included in any project design and authorisation, as an appropriate site selection and design of WEFs can minimize their effects.

Gill (2005) in a recent review of the ecological effects of renewable energy devices in the coastal zone, illustrated the sharp increase in the number of peer-reviewed science articles in this area since the early 1990s. However less than 10% of those articles were related to environmental impacts and even fewer addressed ecological consequences related to the construction, deployment and decommissioning of renewable energy devices. In preparing its response to this request, WGECO also noted the general paucity of peer-reviewed publications particularly with respect to tidal barrages/fences, tidal streams and wave energy devices. Given that these devices have the potential to produce significant near- and far-field effects on coastal ecosystems in particular, and that decision-makers and the public want information on such effects before such energy schemes are implemented, WGECO draws attention to the fact that more scientific research is needed.

9.5 Conclusions

The World needs sources of energy that are low carbon and wet renewables represent a significant resource in the OSPAR region.

Many engineering uncertainties remain but it would appear that with both offshore wave and tidal stream energy the main impacts are associated with habitat change. Provided habitats selected are not rare, there is no reason why, at the likely scale of development, the habitat changes should be seen as significant or preclusive.

Barrages and tidal fences require coastal locations and particular environmental conditions. The ecological consequences are large but can be mitigated and the energy return (and hence carbon emissions reduction) are equally significant.

9.6 Requested Advice from WGECO

- Tidal barrages in locations where they will generate significant levels of power will alter tidal processes over large areas (potentially regional sea scales) although there is scope for mitigation of many of the direct ecological impacts. Many of the sites suitable for use will be RAMSAR sites. While turbine life may be of the order of two decades the barrage structure will potentially have a design life of >100 years.
- Tidal stream devices to generate significant power output will occupy large areas of sea for several decades. Although devices are likely to be well spaced within a farm, the sites themselves will have a large spatial footprint. Adoption of effective marine plans by Member States and within Regional Seas will be necessary to address this concern.
- Wave energy collectors have the potential to alter water column and sea bed habitats and by changes in the wave environment cause changes some distance from the installation. The scale of the impacts is limited and will scale with the size of development and vary depending on the nature of the location selected. Effective marine spatial planning and rigorous licensing requirements will do much to minimise the possible environmental impacts. Most effects would be reversible, fairly rapidly, if an installation was removed.
- Tidal barrages represent a major modification to the coastal environment impinging on natural processes, including bird feeding areas and the migration routes of catadromous and anadromous fish and many maritime

sectors. These changes need to be balanced against the potential to deliver very significant quantities of low carbon energy. The scale of the construction projects for barrages and fences is potentially large and many of the major impacts associated with this phase, for example noise from pile driving, can be mitigated by careful planning, for example by avoiding critical times of year for marine mammals.

- Tidal stream devices and wave energy collectors themselves will have generally only local impacts, similar to those already encountered during routine marine construction activities. Potential concerns with impacts to pelagic organisms still need to be resolved, but are not considered a serious threat at this stage.
- The fact that wave energy and tidal stream devices are still in the experimental/trail phases means that there is no data on the environmental effects of commercial developments. Appropriate scientific studies should therefore accompany the licensing of the first commercial scale installations.

9.7 References

Anon. 2008. Roosevelt Island Tidal Energy Project FERC No. 12611., Draft Kinetic Hydropower Pilot Licence Application Volume 2 of 3. November 2008.

Awatea. 2008. Environmental Impacts of Marine Energy Converters, Report prepared by Power Projects Limited in association with the National Institute of Water and Atmospheric Research. 7 November 2008.

Balmford, A., Gravestock, P., Hockley, N., McClean, C. J. and Roberts, C. M. 2004. The worldwide costs of marine protected areas. Proceedings of the National Academy of Sciences of the United States of America, 101: 9694–9697.

Bochert, R. and Zettler, M. L. 2004. Long-term exposure of several marine benthic animals to static magnetic fields. Bioelectomagnetics, 25: 498–502.

Boehlert, W. G., McMurray, G. R. and Tortorici, C. E. 2007. Ecological Effects of Wave Energy Development in the Pacific Northwest. A Scientific Workshop, October 11–12. NOAA Technical Memorandum NMFS-F/SPO-92.

Bondi, H. 1981. Tidal power from the Severn Estuary"-Volume 1. Energy Paper 46, HMSO, London.

Carstensen, J., Henriksen, O. D. and Teilmann, J. 2006. Impacts of offshore wind farm construction on harbour porpoises: acoustic monitoring of echo-location activity using porpoise detectors (T-PODS). Marine Ecology Progress Series, 321.

CMACS. 2003. A Baseline Assessment of Electromagnetic Fields Generated by Offshore Windfarm Cables, Centre for Marine and Coastal Studies: COWRIE Report EMF-01-2002 66. http://www.offshorewind.co.uk (accessed August 20, 2008), Liverpool, UK.

Coutant, C. C. and Whitney, R. R. 2000. Fish Behavior in Relation to Passage through Hydropower Turbines: A Review. Transactions of the American Fisheries Society, 129: 351–380.

Craig, C., Wyllie-Escheverria, S., Carrington, E. and Shafer, D. 2008. Short-term sediment burial effects on the seagrass *Phyllospadix scouleri*, EMRPP Technical Notes Collection (ERDC TN-EMRRP-EI-03). Vicksburg, MS: U.S. Army Engineer Research and Development Center.

Davis, N., VavBlaricom, G. R. and Dayton, P. K. 1982. Man-made structures on marine sediments: effects on adjacent benthic communities. Marine Biology, 70: 295–303.

DFO. 2009. Assessment of Tidal and Wave Energy Conversion Technologies in Canada, DFO Canada, Scientific Advisory Secretariat. Scientific Advisory Report 2009/064, Ottawa.

Durif, C. M. F., Phillips, J., Browman, H. I., Vollestad, A., Skiftesvik, A. B. and Stock-Hausen, H. *in preparation*. Role of the Earth's geomagnetic field in the short and long distance migrations of the European eel.

Falcão, A. F. D. O. 2010. Wave energy utilization: A review of the technologies. Renewable and Sustainable Energy Reviews, 14: 899–918.

Finstad, J. L. and Noreide, J. T. 2004. Acoustic repertoire of spawning cod, *Gadus morhua*. Environmental Biology of Fishes, 70: 427–433.

Gill, A. B. 2005. Offshore renewable energy: ecological implications of generating electricity in the coastal zone. Journal of Applied Ecology, 42: 605–615.

Gill, A. B. and Taylor, H. 2001. The potential effects of electromagnetic fields generated by cabling between offshore wind turbines upon Elasmobranch Fishes, Countryside for Wale Report 488:73, Bangor.

Gordon, J., Gillispie, D., Potter, J., Frantzis, A., Simmonds, M. P., Swift, R. and Thompson, D. 2004. The effects of seismic surveys on marine mammals. MarineTechnology Society Journal, 37: 16–34.

Gray, A. J. 1992. The ecological impact of estuarine barrages. p. 46. British Ecological Society/Field Studies Council, Shrewsbury.

Hauge, M. 2010. Det ål-timate navigasjonssystemet. Short note, forskning.no. March 10., 2010. http://www.forskning.no/artikler/2010/mars/244097.

Hawkins, A. D. and Amorima, M. C. P. 2000. Spawning sounds of the male haddock, *Melanogrammus aeglefinus*. Environmental Biology of Fishes, 59: 29–41.

Henriksen, O., Carstensen, J., Tougaard, J. and Teilmann, J. 2004. Effects of the Nysted Offshore Wind Farm construction on harbor porpoises: Annual status report for the acoustic TPOD monitoring programme during 2003, http://www.wind-energie.de/fileadmin/dokumente/Themen_A-Z/Offshore/report-nysted_environmental2002.pdf.

Hui, C. A. 1994. Lack of association between magnetic patterns and the distribution of free-ranging dolphins. Journal of Mammalogy, 75: 399–405.

Jenner, H. A., Whitehouse, J. W., Taylor, C. J. L. and Khalanski, M. 1998. Cooling water management in European power stations Biology and control of fouling. Hydroécological Applications, 10: 1–225.

Kaiser, M. J. 2005. Are marine protected areas a red herring or fisheries panacea? Canadian Journal of Fisheries and Aquatic Sciences, 62: 1194–1199.

Kalminjn, A. T. 1982. Electric and magnetic field detection in elasmobranch fishes. Science, 218: 916–918.

Kogan, I., Paull, C. K., Kuhnz, L. A., Burton, E. J., Thun, S. V., Greene, H. G. and Barry, J. P. 2006. ATOC/Pioneer Seamount cable after 8 years on the seafloor: Observations, environmental impact. Continental Shelf Research, 26 771–787.

Langhamer, O. 2010. Effects of wave energy converters on the surrounding soft-bottom macrofauna (west coast of Sweden). Marine Environmental Research, *In press*.

Langhamer, O., Haikonen, K. and Sundberg, J. 2010. Wave power-Sustainable energy or environmentally costly? A review with special emphasis on linear wave energy converters. Renewable and Sustainable Energy Reviews, 14: 1329–1335.

Langhamer, O. and Wilhelmsson, D. 2009. Colonisation of fish and crabs of wave energy foundations and the effects of manufactured holes - A field experiment. Marine Environmental Research, 68: 151–157.

Langhamer, O., Wilhelmsson, D. and Engström, J. 2009. Artificial reef effect and fouling impacts on offshore wave power foundations and buoys - a pilot study. Estuarine, Coastal and Shelf Science, 82: 426–432.

Langlois, T. J., Anderson, M. J. and Babcock, R. C. 2005. Reef-associated predators influence adjacent soft-sediment communities. Ecology, 86: 1508–1519.

Lewis, L. J., Davenport, J. and Kelly, T. C. 2003. A study of the impact of a pipeline construction on estuarine benthic invertebrate communities. Part 2. Recolonization by benthic invertebrates after 1 year and response of estuarine birds Estuarine, Coastal and Shelf Sciences, 57: 201–208.

Lohmann, K. J. and Johnsen, S. 2000. The neurobiology of magnetoreception in vertebrate animals. Trends in Neurosciences, 23: 153–159.

Lohmann, K. J., Putman, N. F. and Lohmann, C. M. F. 2008. Geomagnetic imprinting: A unifying hypothesis of long-distance natal homing in salmon and sea turtles. Proceedings of the National Academy of Sciences of the United States of America, 105: 19096–19101.

Lohse, D. P., Gaddam, R. N. and Raimondi, R. T. 2008. Predicted Effects of Wave Energy Conversion on Communities in the Nearshore Environment. *In* Developing Wave Energy in Coastal California: Potential Socio-Economic and Environmental Effects. . Ed. by P. A. Nelson, D. Behrens, J. Castle, G. Crawford, R. N. Gaddam, S. C. Hackett, J. Largier, D. P. Lohse, K. L. Mills, P. T. Raimondi, M. Robart, W. J. Sydeman, S. A. Thompson and S. Woo. California Energy Commission, PIER Energy-Related Environmental Research Program and California Ocean Protection CouncilCEC-500-2008-083. http://www.resources.ca.gov/copc/docs/ca_wec_effects.pdf (accessed November 19, 2008).

Lugli, M., Pavan, G. and Torricelli, P. 1996. The importance of breeding vocalizations for mate attraction in a freshwater goby with composite sound repertoire. Ethology, Ecology and Evolution, 8: 343–351.

Madsen, PT, Johnson M, Miller PJO, Aguilar Soto N, Lynch J and Tyack P 2006. Quantitative measures of air-gun pulses recorded on sperm whales (*Physeter macrocephalus*) using acoustic tags during controlled exposure experiments. J Acoustical Society of America, 120: 2366–2379.

Marra, L. J. 1989. Sharkbite on the SL submarine lightwave cable system: history, causes and resolution. IEEE Journal of Oceanic Engineering, 14: 230–237.

McCauley, R. D. and Cato, D. H. 2003. Acoustics and marine mammals: Introduction, importance, threats and potential as a research tool. *In* Marine mammals: Fisheries, tourism, and management issues, p. 460. Ed. by N. Gales, M. Hindell and R. Kirkwood. CSIRO Publishing, Collingwood.

Menge, B. A. 1976. Organization of the New England Rocky intertidal community: role of predation, competition, and environmental heterogeneity. Ecological Monographs, 46: 355–393.

Meyer, C. G., Holland, K. N. and Papastamatiou, Y. P. 2004. Sharks can detect changes in the geomagnetic field. Journal of the Royal Society Interface, 2: 129–130.

Misund, O. A. and Aglen, A. 1992. Swimming behaviour of fish schools in the North Sea during acoustic surveying and pelagic trawl sampling. ICES Journal of Marine Science, 49: 325–334.

Murawski, S. A. 2005. Effort distribution and catch patterns adjacent to temperate MPAs. Ices Journal of Marine Science, 62: 1150–1167.

Murawski, S. A., Rago, P. and Fogarty, M. 2005. Spillover effects from temperate marine protected areas. American Fisheries Society Symposium, Tampa, FL.

Paulin, M., G., 1995. Electroreception and the compass sense of sharks. Journal of Theoretical Biology, 174: 325–339.

Pelc, R. and Fujita, R. M. 2002. Renewable energy from the ocean. Marine Policy, 26: 471–479.

Popper, A. N. and Hastings, M. C. 2009. The effect of human-generated sound on fish. Integrative Zoology, 4: 43–52.

Rice, J. C. 2005. Ecosystem Effects of Fishing: Impacts, Metrics and Management Strategies. 187 pp.

Sánchez-Jerez, P., Gillanders, B. M., Rodríguez-Ruiz, S. and Ramos-Esplá, A. A. 2002. Effect of an artificial reef in *Posidonia* Meadows on fish assemblage and diet of *Diplodus annularis*. ICES Journal of Marine Sciences, 59: 59–68.

Simpson, S. D., Meekan, M. G., Jeffs, A., Montgomery, J. C. and McCauley, R. D. 2008. Settlement-stage coral reef fish prefer the higher-frequency invertebrate-generated audible component of reef noise. Animal Behaviour, 75: 1861–1868.

The Engineering Business Ltd. 2005. Stingray Tidal Stream Energy Device - Phase 3, T/06/00230/00/REP URN 05/864. 110 p. + appendices. http://www.engb.com/downloads/Stingray%20Phase%203r.pdf (accessed August 18, 2008).

Tougaard, J., Carstensen, J., Henriksen, O., Skov, H. and Teilmann, H. 2003. Short term effects of the construction of wind turbines on harbor porpoises at Horns Reef., Hedeselskabet Roskilde, Denmark.

Underwood, G. J. C. 2010. Microphytobenthos and phytoplankton in the Severn estuary, UK: Present situation and possible consequences of a tidal energy barrage. Marine Pollution Bulletin, 61: 83–91.

US Department of Energy. 2009. Report to Congress on the Potential Environmental Effects of Marine and Hydrokinetic Energy Technologies, US Department of Energy, Washington D.C.

Westerberg, H. and Begout-Anras, M. L. 2000. Orientation of silver eel (*Anguilla anguilla*) in a disturbed geomagnetic field. *In* Advances in Fish Telemetry. Proceedings of the 3rd Conference on Fish Telemetry, pp. 149–158. Ed. by A. Moore and I. Russell. CEFAS, Lowestoft.

Westerberg, H. and Lagenfelt, I. 2008. Sub-sea power cables and the migration behaviour of the European eel. Fisheries Management and Ecology, 15: 369–375.

Widdows, J. and Brinsley, M. 2002. Impact of biotic and abiotic processes on sediment dynamics and the consequences to the structure and functioning of the intertidal zone. Journal of Sea Research, 48: 143–156.

Wilhelmsson, D. and Malm, T. 2008. Fouling assemblages on offshore wind power plants and adjacent substrata. Estuarine, Coastal and Shelf Science, 79: 459–466.

Wilhelmsson, D., Malm, T. and Ohman, M. C. 2006. The influence of offshore windpower on demersal fish. Ices Journal of Marine Science, 63: 775–784.

Wolf, J., Walkington, I., Holt, J. and Burrows, R. 2009. Environmental Impacts of Tidal Power Schemes. Liverpool, Proudman Oceanographic Laboratory: 18 pp.

World Energy Council. 2007. Survey of Energy Resources 2007, World Energy Council, London.

Öhman, M. C. and Wilhelmsson, D. 2005. VINDREV-Havsbaserad vindkraft som artificiella rev: effekter på fisk. Vindforsk, FOI/Energimyndigheten.

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Annex 2: Agenda

Working Group on the Ecosystem Effects of Fishing Activities (WGECO)

1000 Wednesday 7 April–1700 Wednesday 14 April 2010

Atlantic Room, 4th Floor ICES HQ

H.C. Andersens Boulevard 44–46 Copenhagen

1000 Wednesday 7 April

Plenary

Introductions

Presentation on using ICES Sharepoint/Printer and other services **Helle Gjeding Jørgensen (ICES Secretariat)**

Overview of meeting work plan **Ellen Kenchington (Chair)**

Presentation on WGECO approach to **ToRs a/b**: Assess the development of integrated ecosystem assessments/ Data analyses required to examine the relationships between perturbation and recovery capacity, or some other element of "cost" **Marie-Joelle Rochet/Jake Rice**

Presentation on WGECO approach to **ToR g**: Environmental interactions of wave and tidal energy generation devices **Chris Frid**

Presentation on WGECO approach to **ToR d/e**: Review methods used to determine "good environmental status" under the WFD, HD and MSFD/Conduct a detailed, quantitative, evaluation of a limited number (2 or 3) of management schemes **Jake Rice**

Discussion groups for ToRs a/b, g, d/e; Uploading material to Sharepoint

900–1000 Thursday 8 April

Meeting of ToR leaders for ToRs a/b, d/e, f to inform one another of direction each group is taking

Discussion groups for ToRs a/b, g, d/e

1500

Plenary

Presentation on WGECO approach to **ToR c**: Large fish EcoQO indicator **Simon Greenstreet**

Presentation on WGECO approach to **ToR f**: Extending marine assessment and monitoring framework **David Reid**

Presentation by **Dominic Rihan** on potential ToRs in common with the gear technology WG

Discussion groups for ToRs c, f

900 Friday 9 April

Discussion groups for all ToRs

*****Meeting to follow a format of break-out group and plenary discussion as required with times to be posted daily based on progress *****

Weekend: WGECO works through both Saturday and Sunday with a later start on Saturday and a late day plenary on Sunday. A group dinner is planned for Saturday night.

Wednesday 14 April

The last plenary session will be scheduled for the morning. The afternoon will be spent tidying up the Report, finalizing references, etc. Each ToR group should identify at least one member who will be present Wednesday afternoon to do this. Meeting adjourned 1700.

Annex 3: WGECO terms of reference for the next meeting

The **Working Group on the Ecosystem Effects of Fishing Activities** [WGECO] (Chair: D. Reid, Ireland) will meet in Copenhagen, Denmark from 13–20 April 2011 to:

a) Provide guidance on the use of the proportion of large fish indicator in areas outside of the North Sea;

b) Review the use of science in the development and implementation of "integrated ecosystem management plans" (IEMPs) including objectives setting and performance evaluation as well as other considerations.

WGECO will report by DATE to the attention of the Advisory Committee.

Supporting Information

Priority:	The current activities of this Group will lead ICES into issues related to the ecosystem affects of fisheries and other human pressures on marine ecosystems. Consequently, these activities are considered to have a very high priority.
Scientific justification and relation to action plan:	Action Plan No: 1. Term of Reference a) The Proportion of Large Fish Indicator (LFI) was developed over some years by WGFE and WGECO, and based principally on North Sea trawl survey data. Given that the LFI metric is now recommended in the DCF, and is being implemented outside the North Sea, WGECO recognises the need to provide guidance on how the "tuning" of the LFI was done in the North Sea, and on how it might be done in other sea areas. Critical areas to address would be: Choice of species suite – e.g. pelagic species were not used in the North Sea, but this may not be appropriate elsewhere; Choice of cut-off level. Extensive study suggested the 40cm level, but again this may well be incorrect in other ecosystems; Choice a weight based or number based metric. In the North Sea the metric is weight based, but number based approaches were also considered. Term of Reference b) In addressing ToR e) of the 2010 meeting WGECO proposed a contiuation of the ToR in 2011 specifically to address the following points: 1. Information availability and objectives setting: The intent of this review /discussion would be to inform governance processes of the scale of investment in science that is necessary to deliver different scales and specificities of objectives and management measures. It may also help to manage expectations for the level of specificity in objectives and management measures that can be realized, given an idea of the amount and types of information that be made available. 2. Importance of science support in different stages of IEMP development: Reviewing and summarizing how science demands were met before, during, and after developing of the IEMPs. Are there practices that increase the burden on science to support their development, that decrease the burden, and at least increase value of the science that was input to the process? With regards to science support for the further steps in the process to fully implement IEMPs: 3. Importance of inter-agency communication for IEMP implementation efficiency: Implementation of the IEMPs will require coordinated action by many agencies, with differences in mandates, priorities, risk tolerances, and operational cultures. 4. Criteria for evaluating IEMPs: At some point the consequences of IEMPs will have to be evaluated; one evaluation has already been completed for the Barents sea plan. The evaluations are likely to include social and economic consequences outside ICES traditional scope of activities. However the ecological consequences will have to be evaluated, likely both with regard to progress towards objectives and scale of benefits relative to scale of costs. A review next year around these general questions could contribute to more efficient use of science in the development and implementation of IEMPs, and to IEMPs being both stronger and more likely to produce the desired benefits.
Resource requirements:	The research programmes which provide the main input to this group are already underway, and resources are already committed. The additional resource required to undertake additional activities in the framework of this group is negligible.

Participants:	The Group is normally attended by some 20–25 members and guests.
Secretariat facilities:	None.
Financial:	No financial implications.
Linkages to advisory committees:	WGECO reports to ACOM.
Linkages to other committees or groups:	There is a very close working relationships with WGFE and WGFTB.
Linkages to other organizations:	The work of this group is closely aligned with similar work in FAO and is current with ecosystem approaches under development with the EU, OSPAR, NEAFC and NAFO.

Annex 4: Recommendations

We suggest that each Expert Group collate and list their recommendations (if any) in a separate annex to the report. It has not always been clear to whom recommendations are addressed. Most often, we have seen that recommendations are addressed to:

- Another Expert Group under the Advisory or the Science Programme;
- The ICES Data Centre;
- Generally addressed to ICES;
- One or more members of the Expert Group itself.

RECOMMENDATION	FOR FOLLOW UP BY:
1. WGECO recommends that WGFE be made aware of the suggested ToR a) for 2011 which will continue work on the proportion of large fish indicator.	WGFE
2.	
3.	
4.	
5.	
6.	

After submission of the Report, the ICES Secretariat will follow up on the recommendations, which will also include communication of proposed terms of reference to other ICES Expert Group Chairs. The "Action" column is optional, but in some cases, it would be helpful for ICES if you would specify to whom the recommendation is addressed.

Annex 5: Technical minutes from the Energy Review Group

- RGENG
- By correspondence 10 May 2010
- Participants: Howard Platt (UK Chair), Jakob Asjes (Netherlands), Antonio Sarmento (Poland), Lars Bie Jensen (Denmark), Claus Hagebro and Michala Ovens (ICES Secretariat)
- Working Group: WGECO

Special request for advice from OSPAR June 2009: ICES 4–2010

Request

To provide advice on the extent, intensity and duration of direct and indirect effects and interactions of marine wet renewable energy production (wave, tidal stream and tidal barrage systems) with the marine environment and ecosystems of the OSPAR maritime area, and with pre existing users of these ecosystems, including:

a) actual and potential adverse effects on specific species, communities and habitats;

b) actual and potential adverse effects on specific ecological processes;

c) irreversibility or durability of these effects.

ICES requested advice Review Group summary

The Reports produced by the two ICES Working Groups (WGECO and WGICZM) are, in general, very useful as guidance or consulting documents on environmental assessment or environmental coastal management as regards human activities such as wave and tidal energy deployments.

The WGICZM Report does not fully address the OSPAR request as regards environmental effects (see Annex) but does address the issue of effects on pre-existing users. It is a good review of the potential of the technology to contribute to renewable energy needs.

The WGECO Report, in general, is more comprehensive and covers most of the relevant subjects and the Advice requested from OSPAR as regards potential adverse effects is very complete. There is a paucity of real measurements and data related to the deployment of this technology, so the WG could understandably go no further for the most part than "expert opinion", albeit well-based opinion. The exception is that it should be possible, as stated in the report, to use experience from the offshore wind sector as some of the issues should be very much the same for wet renewables.

Although OSPAR did not request information on windfarms, WGECO make the valid points that:

- These schemes [windfarms] provide useful sources of information on environmental interactions that will be common across all (or most) schemes or that can be extrapolated to provide predictions for the effects of other types of scheme.
- The environmental impacts of electromagnetic emissions from cables, switch gear and sub-stations is the same irrespective of the energy generating device and thus the lessons learnt from offshore wind power develop-

ments are applicable to developments harnessing tidal stream or wave energy.

WGECO reviewed the available evidence from the three requested technologies:

- barrages or fences across coastal bays, fjords or estuaries;
- tidal stream energy in tidal currents; and
- energy associated with waves.

Understandably, Advice from the WGECO on the **actual** effects of wet renewables are fewer than more general statements on the advantages and potential for this technology and what might **potentially** be the effects on the marine environment and ecosystems.

Valid general points are

- To date tidal fences, tidal stream farms and wave energy capture devices have only been deployed on an experimental scale and so prediction of their impacts is based on very limited empirical data.
- At this time, there are insufficient data to state definitively how fish and fish habitat will be impacted by the operation of tidal stream power projects.
- Further work is needed on the possibility that diving birds may respond to the moving blades as potential prey and be attracted to their vicinity.
- Decision-makers and the public want information on such effects before such energy schemes are implemented, WGECO draws attention to the fact that more scientific research is needed.

Specific points are

Barrages or fences across coastal bays, fjords or estuaries

Unsupported but possibly cogent opinions:

- The presence of a barrage also influences habitats upstream and downstream of the facility. This has potential negative implications for birds, although this effect occurs as the flats above the barrage become exposed.
- The implications for tidally feeding fish are the opposite to those of the birds with greater periods for foraging available due to the retention/raising of water levels.
- Changed spatial flow patterns will result in altered patterns of sediment deposition and movement. This will have impacts on benthic communities.
- Changes in the nature of the habitats will alter their suitability as nursery or spawning areas for fish.
- During the construction phase it is inevitable that the natural benthic habitat will be destroyed and eliminated, but the structures themselves will create artificial habitat for benthic organisms.
- Pile driving would likely affect schooling fish or any species with a swim bladder.

Advice:

- Mitigation using salmon ladders is well developed and proven technology for hydroelectric dams.

- There is room for mitigation through engineering of the fence structure to allow spaces for fish to pass through between the caisson wall supporting the turbines and the rotors.

Tidal stream energy in tidal currents

Unsupported but possibly cogent opinions:

- The installation and operation directly affects benthic habitats by altering water flows, wave structures, or substrate composition.
- Bottom disturbances will result from the temporary anchoring of construction vessels; digging and refilling the trenches for power cables; and installation of permanent anchors, pilings, or other mooring devices.
- Fish and other mobile organisms will be displaced and sessile organisms smothered in the limited areas affected by these activities.
- Species with benthic-associated spawning or whose offspring settle into and inhabit benthic habitats are likely to be most vulnerable to disruption during installation.
- The impact of tidal stream energy generation on the marine environment will be broadly equivalent to that of offshore wind farms in their construction, operation and spatial footprint.
- Although of potential concern, there is little scientific literature to suggest that operation of underwater tidal stream energy devices will cause elevated levels of mortality to pelagic organisms such as fish and marine mammals.
- The impacts of tidal stream farms on seabirds are also reported to be small.
- The effect on transport and deposition of sediment may also influence organisms living on or in the bottom sediments, and plants and animals in the water column.
- Changes in water velocities and sediment transport, erosion, and deposition caused by the presence of new structures will alter benthic habitats.
- Deposition of organic matter in the wakes of tidal farms could encourage the growth of benthic invertebrate communities that are adapted to that substrate. Predation by fish attracted to artificial structures can greatly reduce the numbers of benthic organisms.
- If installation involves pile driving, nearby noise levels are likely to exceed threshold values for the protection of fish and marine mammals.
- Cumulative noise production from large numbers of units has the potential to mask the communication and echolocation sounds produced by aquatic organisms in the vicinity of the structures.
- There are considerable information gaps regarding the effects of noise generated by tidal stream farms on cetaceans, pinnipeds, turtles, and fish.
- Tidal stream farms are unlikely to create magnetic fields strong enough to cause physical damage to marine organisms.

Other potential effects were identified as due to:

- Electrical cables (and movement of them);
- Permanent structures on the bottom;
- Marine fouling organisms;

- Fish avoidance;
- Collision risk of marine mammals with static structures such as tidal stream farms;
- Probability of cetaceans failing to detect and avoid a large static structure is considered to be extremely low.

Advice:

- Feeding and breeding sites in particular for marine mammal species should be avoided when tidal farm sites are selected.
- Unlike wind farms, there may not be much visible infrastructure above the water surface, so hindrance to normal marine navigation will need to be carefully managed.

Energy associated with waves

Unsupported but possibly cogent opinions:

- The author [Langhamer, 2010] concludes that the wave energy converters had only minor direct effects on the benthic community (macrofaunal biomass, densities, species richness and biodiversity) in relation to the natural high variations.
- Regarding the pelagic habitat, buoys have a minimal impact on phytoplankton.
- Wave power plants act as wave breakers, calming the sea, and the result may be to slow the mixing of the upper layers of the sea, which could cause an adverse impact on the marine life and fisheries.
- Maintenance involves the use of service boats regularly, sometimes small boats, but some other times, big boats able to transport devices to a port if needed. It also carries some risk of shipping accident.

Irreversibility or durability of these effects

There is not a great deal of advice on irreversibility and durability. However the following opinions were reported:

- Building the barrage across the bay/estuary will destroy the former habitat in the development footprint.
- Construction and decommissioning will also probably result in impacts to adjacent intertidal areas used for construction of caissons or as staging areas.
- Physical impact from small-scale tidal stream generation pilot projects have been found to be reversible on decommissioning [evidence?], especially as the areas most suitable for tidal power generation are located where high current flow causes natural disturbance to the sediments.
- When construction [tidal stream] is completed, disturbed areas are **likely** to be re-colonized by these same organisms, assuming that the substrate and habitats are restored to a similar state.

Effects on pre-existing users

Both WGs appear to have largely missed the point that OSPAR also asked for advice on extent, intensity, etc. of **effects on pre-existing users.** The following extracts in the WGECO Report pertain:

- The presence of the barrage or fence will result in, probably a 0.5 nautical mile exclusion on either side for fishing vessels, vessels anchoring, etc.
- On most large barrage proposals the passage of shipping through the barrage is maintained by the provisions of appropriate lock systems with associated breakwaters and channels. Thus the effect of exclusions zones is minimal for most users.
- Whilst other human activities are likely to be excluded in the area of marine energy converters arrays, the exclusion zones may create de facto marine reserves, in which marine life can flourish.
- Industrial sectors such as fishing and shipping are likely to have concerns regarding both spatial exclusions around tidal stream farms.
- This potential exclusion will need to be carefully managed to avoid conflicts with other sea users, such as shipping lanes, marine aggregate extraction, etc., where site specific requirements for access may also exist.

The WGECO Report contains a final section specifically addressing the requested Advice. RG comments for discussion are in parentheses

- **Tidal barrages**... will alter tidal processes over large areas (potentially regional sea scales) **[Agree]** although there is scope for mitigation of many of the direct ecological impacts. **[Not convinced that mitigation is possible.]**
- **Tidal barrages** represent a major modification to the coastal environment impinging on natural processes. The scale of the construction projects for barrages and fences is potentially large and many of the major impacts associated with this phase, for example noise from pile driving, can be mitigated by careful planning, for example by avoiding critical times of year for marine mammals. **[Agree]**
- **Tidal stream** devices .... will occupy large areas of sea for several decades. Adoption of effective marine plans by Member States and within Regional Seas will be necessary to address this [environmental] concern. **[Agree]**
- **Tidal stream** devices and wave energy collectors themselves will have generally only local impacts, similar to those already encountered during routine marine construction activities. Potential concerns with impacts to pelagic organisms still need to be resolved, but are not considered a serious threat at this stage. **[Not convinced that this can be stated with such conviction. Depends on definition of 'local impacts' and 'serious threat'.]**
- **Wave energy** collectors have the potential to alter water column and sea bed habitats and by changes in the wave environment cause changes some distance from the installation. ...... Most effects would be reversible, fairly rapidly, if an installation was removed. [**Equivocal: possibly not rapidly reversible if populations of marine mammals have been adversely effected.]**
- The fact that wave energy and tidal stream devices are still in the experimental/trail phases means that there is no data on the environmental effects of commercial developments. Appropriate scientific studies should therefore accompany the licensing of the first commercial scale installations. **[Agree]**

Conclusions in the WG Report

RG did not agree entirely with all of the conclusions of the WGECO Report (Section 9.5 of the Report, page 190).

Many engineering uncertainties remain but it would appear that with both offshore wave and tidal stream energy the main impacts are associated with habitat change.

Disagree: Habitat change is important but there could be more serious adverse impacts on marine mammals and fish.

Provided habitats selected [for offshore wave and tidal stream energy] are not rare, there is no reason why, at the likely scale of development, the habitat changes should be seen as significant or preclusive.

Disagree: Depending on the 'scale of development', effects could well be significant and permanent. In sensitive coastal areas, especially for Natura 2000 and RAMSAR sites, extensive developments may well lead to more significant impacts. Not all, in fact most marine habitats of Community interest would be classified as 'rare'.

The ecological consequences [of barrages and tidal fences] *are large but can be mitigated.*

Disagree: RG believe that impacts will be significant and irreversible and cannot be mitigated for; other by removal of the installation. Evidence from the Eastern Schelt Estuary in the Netherlands is relevant, where the installed storm-surge barrier has resulted in siltation of tidal flats and erosion of salt marshes. Compensatory measures by creating an equivalent ecosystem elsewhere could be considered but in reality this is not usually feasible. Therefore there would need to be recourse to the Over-riding Public Interest (OPI) argument.

Recommendations from the RG

For completeness, additional potential adverse effects and aspects for considerations are listed in the Annex 1 and some suggested examples of pertinent screening questions are listed in Annex II.

That the utilisation of marine energy resources is a new and rapidly growing sector of the marine industry and has great potential is a given. However, there are very limited data allowing environmental impacts to be predicted.

There is a large degree of uncertainty regarding what environmental impacts will result from deployments and thus there is little scientific evidence/confidence to go further in the WGECO Report. More information on this subject will only be possible with investigative monitoring at the deployment phase.

This poses an ecological and ethical dilemma. This is particularly stark as the Habitats Directive has now changed the paradigm from presumption in favour of development to presumption in favour of environmental protection. Regulators and developers now need to pay heed to the precautionary principle. However, at the same time, there is almost overwhelming pressure to pursue sustainable renewable energy sources that will not pass on to our later generations intolerable burdens.

So the question is: **How can we develop wet renewables without the scientific evidence base to assess risk?**

Perhaps still controversial, and potentially in conflict with the precautionary principle, it is suggested that the Advice Drafting Group consider advising an **Adaptive Management Strategy** for new installations. This might apply at least until enough field data and experience is amassed from several actual installations to be able to produce more robust environmental impact assessments for proposed developments.

Essentially, this requires allowing a heavily conditioned licence or consent under national legislation at the same time, if appropriate, advising the European Commission if there are habitats and/or species of Community interest that may be potentially subject to deterioration. European case law (the Waddensea judgement) effectively states that **any** deterioration is a significant deterioration.

Each proposed wet renewable installation should be treated for the time being as novel and be dependant on the specific technology proposed and the precise location.

It is advised that any proposal should be subject to detailed initial screening and then if appropriate a Habitats Directive Article 6 assessment. Note that some marine species listed in Annex II are also on Annex IV, with the obligation to provide strict protection wherever they occur, i.e. not just within designated SAC sites.

In trying to address the OSPAR request, and although based on expert knowledge, it might be useful to consider an assessment matrix with impacts classified on predicted levels of irreversibility, durability and extension (near field/far field extension). This will help the definition of impacts prioritization and on the establishment of research priorities regarding environmental monitoring/assessment.

Any proposal under the scheme outlined above should, subject to ADG consideration, provide an analysis of all of the following five phases:

- **Pre-installation:** at least 12 months environmental monitoring of the appropriate marine attributes (including biological, hydrological and physico-chemical) to act as a reference baseline for subsequent phases.
- **Installation**: including effects of any plant needed to install such as Jack-up Barge and effects of sediment caused by drilling to install piles.
- **Commissioning**: deploy marine mammal observers (MMOs), prove active sonar and demonstrate fast shut-down.
- **Operation**: including continued environmental monitoring and maintenance operations.
- **Decommissioning**: including subsequent restoration where appropriate.

Although the focus of the present discussion is on the Reports' review, there are some additional thoughts that might be considered by the ADG. From the developer's point of view, marine energy prototypes are very expensive projects. The environmental concerns are usually considered as non-technological barriers, which can greatly increase project costs, particularly as extensive monitoring will be required. Thus, at this early stage a balanced approach, between scientific, legislative and industry interests is required to optimize effort.

It is still unclear how fast these developments will be and how soon they will provide meaningful amounts of energy. On wave energy, it could be five years for the technology to become pre-commercial and the first 20 MW farms (area 1 km^2) to be deployed. Tidal stream could be somewhat faster. The European Association of Ocean Energy estimates that by 2020 as much as 3.5 GW could be deployed in Europe, this representing a total area of about 175 km^2. This means that with appropriate environmental monitoring of the first (and small) ocean energy farms to be built we will have time to learn and adjust legislation.

The development of ocean energy is being undertaken by small companies – the Carbon Trust report from 2006 refers to £10 M to develop a prototype and £10 M to run a two year sea trial program. These values are optimistic as indicated by developers at a recent Ocean Energy conference in Brussels. At that meeting, Aquamarine reported that they spent £70 M to built and deploy at EMEC their 360 MW prototype.

Licensing can also be very expensive, in part because of the extensive baseline studies and mitigation measures that will be required. Wave Dragon reported a cost of £0.5 M to license their 5 MW prototype in the UK.

It seems reasonable that if society asks companies to risk their money in developing a new technology, society should also accept a reasonable and limited environmental risk. With only small scale projects, say with less than five devices in the same site, environmental impacts should only be marginal if the devices are not in a very sensitive environment and very intrusive techniques are not used in the deployment.

Monitoring of ocean farms is the only way to learn what the environment impacts are and these extensive (and costly) environment programs should in part be supported by public funds, not solely left to the companies involved in the development of the technology.

Extra information

Since the OSPAR request (June 2009) the following pertinent information has become available.

Seagen: www.seageneration.co.uk. Documents available include [requires registration to gain full access]:

Environmental Impact Assessment Environmental Impact Study (Non Technical Summary) : This report surmises the findings of the EIA and is available to non registered users.

Environmental Monitoring Programme: This report details the environmental monitoring that is being conducted pre-installation and will be conducted post installation.

Pre-Installation Baseline Report: This report details the environmental monitoring that is has been conducted pre-installation (April 2005 to July 2006).

SeaGen Biannual Report - February 2009: This report details the environmental monitoring that is being conducted.

Environmental Action and Safety Management Plan 2008: This report details the perceived risks during installation, operation and decommissioning and provides proposals for mitigation. HSD Article 6 Assessment of MCT - February 2008.

H M Government (March 2010). Marine Energy Action Plan 2010. Executive Summary & Recommendations. Department of Energy and Climate Change. www.decc.gov.uk

Environmental Effects of Tidal Energy Development: A Scientific Workshop. University of Washington, Seattle, Washington March 22-24 2010. [Final report not yet available]. Workshop Briefing Paper available at http://depts.washington.edu/nnmrec/workshop/docs/Tidal_energy_briefing_paper.pdf

Also:

Wilson, B., Batty, R. S., Daunt F., and Carter, C. 2007. Collision risks between marine renewable energy devices and mammals, fish and diving birds. Report to the Scottish Executive. Scottish Association for marine science, Oban, Scotland, PA37 1QA.

Annex I: Additional potential adverse effects and aspects for considerations

- Landfall impacts.
- Barrier to juvenile fish using estuaries and coastal areas as a nursery.
- Alien species introductions from installation plant.
- Impact of anti-fouling compounds.
- Potential accidental release of pollutants e.g. lubricants during routine maintenance.
- Effects should also be considered in combination.
- Use of oil-based drilling muds.
- Effects of cuttings if not removed from site for transport to safe disposal.
- Risk analysis based on key questions (see Annex).
- Landscape and visual amenity.
- Engagement and liaison with local people (Aarhus Convention) e.g sailing clubs, anglers.
- Pre-installation baseline monitoring and assessment of existing conservation status (favourable, inadequate, poor).
- Monitor noise level of turbines in operation.
- As regards Europe, advice should be given in the context of the Habitats and Species Directive (HSD) including assessment of alternatives.
- Contingency plan for adverse weather during installation.
- Close collaboration between regulator and developer.
- Avoid breeding seasons and over-wintering migratory bird sites.
- Marine archaeology.
- Restoration of on-shore contractor compound.

Annex II: Examples of key questions for a risk analysis

KEY QUESTIONS	YES/NO
Is the location in or likely to affect a Natura 2000 site?	
Are there habitats or species of Community interest in or near the location of the installation?	
Is marine mammal density and behaviour in the location significantly modified by the installation?	
Does the installation have a significant effect on seal movements through the location?	
Are seals significantly excluded from foraging habitat or social areas within the location as a result of the installation?	
Does operation of the the installation have a significant effect on marine mammal sightings within the immediate waters of the installation?	
If deployed, how far way can the active sonar system detect marine mammals?	
Can the turbine stop before the travel path of a detected marine mammal brings it into a zone of possible injury?	
Does marine mammal activity increase or decrease during night time?	
For all recorded stranding events, have any marine mammal mortalities occurred as a consequence of physical interaction with the installation?	
Does the installation displace cetaceans from the location?	
Does the installation present a significant barrier effect to the free passage of cetaceans through the location?	
Has the number of seal adults and pups decreased significantly within the location?	
Does the installation have a significant effect on seal movements through the location?	
Has there been a significant change in the use of seal haul-out sites in the vicinity of the location?	
Does the installation present a barrier effect to the free passage of seals through the location?	
Does the installation present a barrier effect to the free passage of fish through the location?	
Is there a significant change in the broad benthic community structure that can be attributed to the installation?	
Is there a significant change in abundance of dominant characterising benthic species that can be attributed to the installation?	
Does the installation have a significant impact on seabird activities in the location?	
Does the installation displace foraging diving birds from important areas within location?	
Has the installation significantly modified the flow dynamics, scour patterns or turbulence character of the location?	
If changes in the flow dynamics, scour patterns or turbulence do occur, have they caused a change in benthic community structure and function?	

Annex 6: Technical minutes from the Review Group for OSPAR request on extending marine assessment and monitoring framework used in Ch. 10 of the OSPAR QSR

- RGQSR
- By correspondence 3 May 2010
- Participants: Eugene Nixon (Chair), Thorsten Blenckner (Sweden), Chris Karmann (Netherlands), Claus Hagebro and Michala Ovens (ICES Secretariat)
- Working Group: WGECO

General remarks

A thoroughly argued, in depth advice, mainly focussed on responding to the OSPAR request on how to improve the thresholds, extension to support plankton communities and scale issue.

The document gives a strong impression of being technically correct and contains no substantial flaws, although references to published work is lacking.

Review the methodology used by the OSPAR workshop on the development of Chapter 11 of the QSR 2010 (Utrecht Workshop)

WGECO provides a comprehensive review by providing a good overview what is missing and needs to be considered by OSPAR. Overall, it would be useful to have a section including clear suggestions and recommendations, as those are partly distributed in the entire text.

WGECO notes in paragraph 8.3.3 that the choice of components and the state indicators to be used to represent them should be based on an objective process as described in earlier chapters of the WGECO Report. While reported in the Section on extension of the method to plankton communities, it is a generic remark on the method. Another example can be found in paragraph 8.4.3, last bullet, which deals with the lack of data to support the methodology. Although introduced in the discussion on scale issues, it is a general remark on the methodology.

WGECO states that habitats have no realistic recoverability in ecological time. This is not necessarily true. For example: the habitat “shallow sand bank” is disturbed by abrasion due to fisheries activity. Since it does not affect the ‘principle structure’ of the habitat (the shallow sand bank), the affected part of the habitat may easily recover. Perhaps WGECO could provide a better explanation of their statement. If the habitat was, for example, impacted by sand extracting, affecting the ‘principle structure’ of the habitat, then indeed recovery is unlikely.

Advice on: a) improvements that could be made to the thresholds between different assessment classes, including any scientific basis for proposed thresholds

The issue of arriving at scientific sound thresholds for the assessment classes is extensively discussed in paragraph 8.2 of the WGECO document. In the first paragraph after the introduction (paragraph 8.2.2) a discussion is presented on what *might* be done when setting thresholds for fish, with the comment that it should *not* be considered a recommendation. This appears inconsistent with the objective of giving advice to OSPAR.

In the above mentioned discussions, *inter alia*, comments are made to assumptions of the methodology (changes in relation to 'former natural conditions'. These are useful, but should better be placed in the suggested paragraph containing a review of the method.

Paragraph 8.2.5 provides clear advice on how to arrive at scientifically sound thresholds for the assessment classes. Section 8.2.5.2, page 169, however discusses reference levels on some equivalent of 'sustainable use' and the relation between a perturbation and recovery capacity. This is a complex paragraph and could have been better explained possibly by way of an example.

Finally, in paragraph 8.2.6, conclusions are drawn, assuming that the OSPAR QSR was to be the science basis for implementation of the MSFD, this may (but not necessarily) have biased the conclusions.

Advice on: b) extending the methodology to support the assessment of plankton communities

Clear advice, consistent with earlier analysis on thresholds.

Advice on: c) improving the method for working at different scales

WGECO starts in paragraph 8.4.2 with an elaboration on information sources relative to scale considerations. Following the logic in this paragraph the uncertainty in science advice because of data limitations is not at all related to the scale issue, but inherent to the data availability in a region. For example, data availability is generally poorer in the Mediterranean, while the scale of the assessment is comparable to the North Sea.

It is stated that the method will only identify pressures that are causing impact at the scale of the region chosen for the impact assessment. This is only true for ecosystem elements that are found all over that region. If an ecosystem element occurs in only a small part of the region (e.g., a few kilometres of rocky shore), the regional assessment for that element will be based on the small area in which it occurs. This was true for only a few of the areas assessed during the Utrecht workshop, in general the WGECO statement indeed applies.

The final bullet on page 177 stipulates that a sensitivity analysis would be useful, for example based on a boot strapping analysis. That would be true if the assessment was carried out on a complete set of data, but since it is based on expert opinion this type of sensitivity assessment cannot be carried out. It is, however, possible to perform an uncertainty assessment using the uncertainty scores provided in the audit trail by the experts in the Utrecht workshop -such analysis has not been carried out and would be a very useful exercise.

WGECO conclude that the methodology is applicable to all spatial scales.