Advisory Committee on Ecosystems

ICES CM 2002/ACE:03
Ref. D, E, G

REPORT OF THE

WORKING GROUP ON ECOSYSTEM EFFECTS OF FISHING ACTIVITIES

ICES Headquarters
17–26 March 2002

International Council for the Exploration of the Sea

Conseil International pour l'Exploration de la Mer

Palægade 2–4 DK–1261 Copenhagen K Denmark

TABLE OF CONTENTS

1 OPENING OF THE MEETING

The Working Group on Ecosystem Effects of Fishing Activities (WGECO) met at ICES Headquarters, Copenhagen, from 18–27 March 2002. Attendance at the meeting comprised:

Philippe Archambault	Canada
Niels Daan	Netherlands
Andrey Dolgov	Russia
Martin Dorn	USA
Lisette Enserink	Netherlands
Chris Frid (Chair)	UK (England and Wales)
Simon Greenstreet	UK (Scotland)
Mikko Heino	Norway
Louize Hill	Portugal
Simon Jennings	UK (England & Wales)
John Joyce	Ireland
Ellen Kenchington	Canada
Robert Mohn	Canada
Gerjan Piet	Netherlands
Jake Rice	Canada
Stuart Rogers	UK (England and Wales)
Sigma Steingrimsson	Iceland
Mark Tasker	UK (Scotland)

Contact details are given in Annex 1.

The Working Group members were welcomed by ICES General Secretary, Mr David Griffith. The Terms of Reference for the WGECO meeting were discussed on the first morning, and a plan of work was adopted for the meeting. Special efforts were made to make the fullest progress possible on the Terms of Reference directly supporting the advisory tasks of ICES, and to provide appropriate recommendations for the further development of work in support of those Terms of Reference where the process of arriving at a solution was begun.

Terms of Reference (C. Res 2001/2ACE02) for the meeting were:

The **Working Group on Ecosystem Effects of Fishing Activities** [WGECO] (Chair: Chris Frid, UK) will meet from 18–27 March 2002 at ICES Headquarters to:

a) continue the work started in 2001 to develop the scientific components needed for provision of scientific advice required by an EcoQO framework;

b) quantify the relative role of fishing activity on dynamics of the marine ecosystem and nutrient turnover, in comparison with other comparable human activities such as marine disposal, and mineral extraction, to the extent possible;

c) continue the work plan to test hypotheses about which components of the marine ecosystem are most sensitive to bottom fishing impacts;

d) in response to the EC DG Fish request for an "evaluation of the impact of current fishing practices on non-target species, … and suggestions for appropriate mitigating measures", investigate ways to use data products produced by the Study Group on Discard and By-catch Information for ecosystem management studies [contingent on discard and by-catch from SGDBFI being available for further analyses]. Where data are sufficient, evaluate the impact of fishing on non-target species. Identify species and fisheries where mitigative actions may be warranted and, in such cases, propose and justify alternative mitigation measures;

e) drawing on material compiled by SGCOR, summarize all available information on the distribution of cold-water corals in the ICES area. Based on experience from the ICES area in particular, and more generally from cold waters of northern, southern, and deep-sea areas of the world, relate, to the extent possible, the information on the distribution of corals in the ICES area to threats from fishing activities and other potential disturbances [EC DG Fish];

f) consider the report of the former Planning Group on Comparing the Structure of Marine Ecosystems in the ICES Area and specifically advise on the areas to be used in ecosystem comparisons and the meta-data available for such comparisons;

g) propose a process to be able to summarize available information on the distribution of other sensitive habitats in the ICES area, and evaluate the adequacy of the information as a basis for scientific advice for an "evaluation of the impact of current fishing practices on … sensitive habitats, and suggestions for appropriate mitigating measures"; this should include the definition of criteria or standards for determining what is a "sensitive habitat";

h) propose a process to be able to obtain information to develop advisory forms appropriate to the preservation of genetic diversity, beginning with the initiation of an evaluation of the advisory forms and management approaches that would be necessary and sufficient for the protection of genetic diversity of exploited stocks, and stocks suffering substantial mortality as by-catch;

i) propose a process to be able to obtain information to consider "ecological dependence in management advice, firstly addressing the groups of species with the ecological linkages that are known with high reliability to have strong ecological linkages", including specification of the data requirements and models that would be required to provide the scientific basis for a response to that request. Propose a workplan and timetable for ICES to prepare itself for developing that scientific advice;

j) review progress of activities initiated in 2001 by the Planning Group for a Workshop on [Top-down] Ecosystem Modelling.

An additional term of reference was added shortly before the meeting, based on a new OSPAR request:

k) provide an assessment of the data on which the justification of the habitats in the OSPAR Priority List of Threatened and Endangered Species and habitats will be based; this assessment should be to ensure that the data used for producing the justification are sufficiently reliable and adequate to serve as a basis for conclusions that the habitats concerned can be identified, consistently with the Texel-Faial criteria, as requiring action in accordance with the OSPAR Strategy on the Protection and Conservation of the Ecosystems and Biological Diversity of the Maritime Area.

WGECO will report by 12 April 2002 for the attention of the ACE and the Marine Habitat, Living Resources, and Resource Management Committees.

The timing of the meeting this year allowed WGECO to overlap with the Fifth International Conference on the Protection of the North Sea (the Bergen Conference) and the meeting of the Working Group on the Application of Genetics in Fisheries and Mariculture (WGAGFM). This allowed us to fully integrate the content of the Bergen Declaration, and in particular, the adoption of a pilot EcoQO scheme, into our consideration of Term of Reference (a). In developing our thinking in respect of Term of Reference (h) we were able to use e-mail contact to seek and receive advice from WGAGFM during the course of their meeting.

Acknowledgements:

The Working Group would like to thank Marianne Neldeberg for excellent assistance, patience, and good humour in supporting our computing, system networking, and data requirement, and Louise Scharff and Vivian Piil for general logistical support and untiring assistance in a diversity of areas. It also extends thanks to Dr Jan van Delfsen (Netherlands) and Dr Siân Boyd (UK) for providing information to the group on marine aggregate extraction that would otherwise have been unavailable. WGAGFM provided invaluable help and support at the end of an e-mail link; we wish to thank them for including consideration of our activities into their already full work programme.

2 EXECUTIVE SUMMARY

This was the first meeting of WGECO on an annual cycle. We adopted this pattern to make it easier to provide timely outputs in support of ICES advisory work, and balance the workload between requests for this and the need to advance basic and applied knowledge. Once again, our workload contained a mix of requests for work in support of advice, intellectual development of work-plans to underpin future requests for advice, and original research. The Chair would like to record his thanks to all the members of WGECO for their hard work and commitment both during the meeting and in preparing for it.

Much of the workload in 2001 concerned the development and application of EcoQOs and in **Section 3** we update this with a consideration of the developments that have occurred, primarily through OSPAR and the North Sea Ministerial Conference, since then. We provide some general commentary, and then focus on the work that needs to be done to provide the science base to make this approach operational following the Bergen Declaration. This work is considered in terms of short-term, urgent, actions that are required to implement the pilot scheme and medium-term activities focused on the remaining OSPAR list of EcoQ elements associated with the 10 issues. We then consider the longer-term and more generic considerations that need to be addressed in recognition that advancing the EcoQ – EcoQO framework for the 10 OSPAR issues is, itself, only a start in the process of the adoption of the ecosystem approach to environmental management.

Back in 1992, WGECO considered the relative impact of fishing and other human activities on marine ecosystems. Our Term of Reference (b) allowed us to revisit this perennial issue and in **Section 4** we provide a detailed analysis of the extent of beam trawling and dredging in the southern North Sea and some quantification of the mortality benthos suffer as a result. While it is impossible, given the data currently available, to provide a definitive answer as to the relative impacts of various activities on marine ecosystem dynamics, we have advanced our thinking on this and have made recommendations which, if implemented, will take us further forward.

In **Section 5** we continue the development of our understanding of the response of ecosystems to fishing activities. This knowledge is critical in our ability to provide advice on these issues and the development of candidates for ecosystem indicators. This year our work focused on comparisons of the behaviour of various metrics in a number of geographical regions, extending across the North Atlantic from Europe to Canada, and from Portugal in the south to the Barents Sea in the north. These analyses confirm the sensitivity to changes in the ecosystem of metrics based on the size spectra of the fish assemblage. The response of some of the other metrics varied among systems, possibly in response to differences in the dynamics of the various systems, or possibly due to inherent sensitivities in the metrics to different types/structures in the data. This highlights the need for considerably more research into the behaviour of some of these metrics before they can be used in a management context. However, the size-based metrics seem to offer the hope of robust metrics.

Progress on Term of Reference (d) was constrained by the lack of appropriate data. While this was frustrating, in **Section 6** we offer some considerations on the needs of recording programmes for discards of non-target species and make recommendations as to the way ICES can progress this important issue.

One of the most widely cited examples of a habitat potentially under threat from fishing activities is that of cold-water corals. In **Section 7** we provide a summary of the current state of knowledge of their distribution in the ICES area, drawn primarily from the work of SGCOR, while acknowledging that the knowledge base on this is in a continual state of flux. We then go on to look at the actual threat posed by fishing to these biogenic structures, and consider mitigation measures. Action to protect these habitats is likely to be the subject of requests for advice in the near future. This is an area where the knowledge base is growing rapidly and so it is important that the topic is regularly reviewed and advice provided based on the latest information. We provide some recommendations to assist ICES in being prepared to provide that advice when it is requested.

Our Term of Reference (f), **Section 8**, also caused considerable frustration. The appropriate spatial areas or data sources for the use in ecosystem comparisons will vary depending on the nature of the questions and, until these are defined, the consideration of specification of data sets and spatial areas is premature. We provide some commentary on the work of the Planning Group on Ecosystem Comparisons and advance the consideration of spatial issues and the nature of data that might be required.

Three Terms of Reference required us to begin work on issues concerned with the provision of advice under an ecosystem approach to management. This work stems from the continued development of the ecosystem approach to management within the European Union. In September 2001 the European Commission requested that ICES begin to develop advice on: “fishing impacts on non-target species and sensitive habitats”, “the provision of protection for species ecologically dependent on species affected by fisheries” (i.e., those with strong ecological linkages) and “preservation of genetic diversity”. These are recognised as the three most immediate areas where management advice needs to adopt a wider ecosystem approach. The EC, however, emphasises in its request the need for advice about ecosystems to follow the norms of scientific development with regard to hypothesis testing and peer review. These three Terms of Reference begin that process. In each case, we were requested to use our expertise to establish exactly what framework could be applied to rigorously handle the various types of information to be assessed, where the necessary data could be found and how it should then be used. It was never the intention that WGECO would complete work on these issues at this meeting but in each case we have made substantive progress.

In **Section 9**, Term of Reference (g), we provide a matrix classification of sensitive habitats (taken from the OSPAR Threatened and Declining Habitats list) against fishing impacts and consider mitigation measures for each significant impact. We then review the effectiveness of this matrix approach before developing a consideration of a decision-tree approach to decision-making, and how this might be progressed as a model for preparing ICES advice on this issue.

The Convention on Biological Diversity requires the conservation of natural levels of genetic information. Predation has always been a powerful evolutionary force, altering the genetic composition of predated species; fishing is also likely to exert a selective pressure. In **Section 10** we review the evidence for, and postulated effects of fishing on fish populations and then consider the framework for providing advice to address these changes. This framework is then developed in an ICES context to produce a plan for the provision of fisheries advice, including consideration of genetic effects.

Predators clearly depend on their prey, although in many cases they can switch prey should the preferred type become rare. Management of human activities using an "ecosystem approach" requires consideration of such ecological linkages – predators with their prey, species with their competitors, the habitat needs of each group of organisms. The list of such linkages is long but what is critical from a management point of view is which are important or "strong ecological linkages", as these are the ones which need to be protected if ecosystem function and the constituent species are to be conserved. In **Section 11** we develop some initial criteria for assessing linkages for their strength and then consider how management advice may need to take account of different classes of link. We then propose a means of advancing this process using the existing expertise and structures within ICES.

Section 12 briefly reviews the proposal for a Workshop on [Top down] Ecosystem Models. The Planning Group (PGEM) for this proposed workshop was established following a recommendation from WGECO in 1999 and their report was reviewed by WGECO in 2001. We now reflect on the comments of ACE on that report and the correspondence from PGEM members. WGECO remains firm in its belief that such a workshop could greatly increase the range of tried and tested tools we have available for assessing the ecosystem effects of fishing activities and in developing the ecosystem based approach to environmental management.

Section 13 deals not with one of our original terms of reference but rather attempts to address, at short notice, a request for advice to ICES from OSPAR on "threatened and declining species and habitats" in the OSPAR region. This request was directed to the Chair of ACE who subsequently asked WG Chairs to do as much as they could to assemble the necessary information. The specific request to WGECO was to consider the habitats section of the OSPAR request and "*to ensure that the data used is sufficiently reliable and adequate to serve as a basis for conclusions that the species and habitats concerned can be identified... as requiring action in accordance with the OSPAR Strategy on the Protection and Conservation of the Ecosystems and Biological Diversity of the Maritime Area"*. Given the short notice, we were constrained by the information that was readily available or could easily be accessed by working group members. However, we recognised the importance of providing this analysis in order to assist ICES in formulating advice to OSPAR and so gave this work a high priority. We have therefore assessed each habitat and given an opinion on the adequacy of the information base used, and that available, in determining its status with regard to both "threatened" and "declining" in the OSPAR region. This information is presented in a standard format to aid comparison and extraction of information on a particular habitat.

Traditionally our reports contain a more wide ranging section, entitled **Food for thought,** that has provided us with an opportunity to include material and ideas developed at the meeting but not immediately germane to one of the Terms of Reference. This section does not appear in this report, not because we had no ideas to develop but simply the workload provided by our eleven Terms of Reference completely filled our time. In **Section 15 Future Activities and Recommendations** we include a number of recommendations for consideration by ICES. In addition to these broad recommendations, many of our Sections also include specific recommendations for advancing those particular areas of work.

3 SCIENTIFIC ADVICE NEEDED BY AN ECOQ - ECOQO FRAMEWORK

3.1 History and Context

Term of Reference a) states "continue the work started in 2001 to develop the scientific components needed for provision of scientific advice required by an EcoQ – EcoQO framework".

This work took on greater importance and urgency during the WGECO meeting, as the Fifth International Conference on the Protection of the North Sea, comprising North Sea Ministers and members of the European Commission responsible for protection of the environment, adopted the Bergen Ministerial Declaration on 21 March 2002. A number of important provisions of the Bergen Declaration are relevant to the interests and past activities of WGECO. The Declaration is consistent with many science initiatives that WGECO has been promoting throughout its 12 years of activity, and brings into focus the need for much more science activity in these areas at this meeting and in meetings to come.

With regard to this specific Term of Reference, Paragraph 2 of the Bergen Ministerial Declaration agrees to "implement an ecosystem approach to the health of the North Sea ecosystems" and that "management will be guided by the conceptual framework" in Annex 2 of the Declaration. The conceptual framework in Annex 2 goes far beyond the activities within the competence of WGECO (including policy decisions, control and enforcement). However, within the scientific components of the framework, explicit objectives play a central role. Correspondingly, Annex 3 of the Declaration lists a set of Ecological Qualities (EcoQs), Ecological Quality Elements, and Ecological Quality Objectives (EcoQOs) that are an opening step in implementing the commitments to an ecosystem approach that are made in the Bergen Declaration. WGECO has been considering the issue of operational ecosystem objectives during several meetings, and takes this Term of Reference as linked directly to the EcoQ – EcoQO aspects of the much larger "conceptual framework for an Ecosystem Approach". For the remainder of this section, we refer to the "EcoQ – EcoQO framework" as the much more restrictive job of identifying, justifying and using EcoQs, EcoQ elements, and EcoQOs, as presented in Annex 3 of the Bergen Declaration. There are other contributions that WGECO could make to the much larger conceptual framework, but they are outside the material needed to address this Term of Reference.

While applauding the important step forward represented by the Bergen Declaration, WGECO stresses that last year's evaluation of the EcoQ – EcoQO framework found loose ends, loose language, and loose thinking to be pervasive in many documents about EcoQs, EcoQOs, etc. In Section 5 of last year's report, WGECO attempted to provide more systematic rigour and direction to the selection and implementation of EcoQs and EcoQOs, as well as suggestions for clearer terminology. These concerns have been addressed to varying extents, and the terminology and argumentation are improved in the background documentation prepared by OSPAR for the Bergen Meeting (OSPAR, 2002). However, the developments over the past year have increased the importance of strengthening the scientific framework for EcoQs and EcoQOs, and making it more operational. **WGECO still thinks that without substantial improvements in rigour of the EcoQ – EcoQO framework, there is still a risk that the EcoQ – EcoQO framework may achieve no more than past scientific advisory and management frameworks**. Therefore, WGECO welcomes the opportunity to build on the work reported last year, with the more specific focus of the Bergen Declaration.

Throughout Section 5 of ICES (2001a), WGECO provided guidance on a number of aspects of a scientifically sound and operationally effective EcoQ – EcoQO framework for the science and management of marine ecosystems. These points were accepted by the Advisory Committee on Ecosystems (ACE), and comprise much of the material in Section 4 of the 2001 ACE Report (ICES, 2001b). We repeat that guidance here in Section 3.1.1, as a starting point for further development in this report. We do not repeat the rationales for our various conclusions and recommendations, where they were developed adequately in the previous report. Following presentation of that information, we step to the long-term perspective. Without intending to diminish the importance of the commitments in the Bergen Declaration, Section 3.2 gives emphasis to the scope of the job remaining, if the EcoQ – EcoQO framework is really to become a comprehensive framework for protection of the North Sea.

For the EcoQOs actually specified in Annex 3 to the Bergen Declaration, the commitment is to move forward immediately with a pilot project. In Section 3.3, below, we consider how well prepared the scientific community is to proceed immediately with implementation of those specific EcoQOs, and what role ICES and WGECO could have in ensuring rapid but sound progress. Several EcoQ *elements* in Annex 3 to the Bergen Declaration do not have specific EcoQOs, and in those cases the Declaration states the intent to proceed with EcoQOs for them within the next two years. Section 3.4, below, considers the status of scientific knowledge relative to those EcoQ elements, and identifies gaps in the scientific basis for relevant EcoQOs. Where gaps are found, we provide guidance on how to address them and the possible roles for ICES and WGECO. Both sections are quite detailed, as they try to provide clear descriptions of necessary work, but differ in the time scale for the necessary actions. Throughout this entire section, we do our best to avoid second-guessing the choices of EcoQ elements and EcoQOs, and offer constructive suggestions for moving ahead, given Annex 3 of the Bergen Declaration.

3.1.1 Criteria for good Ecological Quality metrics

Deriving from several sources (Anon., 1999; Lanters *et al.*, 1999; Kabuta and Enserinck, 2000; ICES, 2001c, 2001d; Piet, 2001), WGECO identified several key features of EcoQ metrics. These were explicitly identified as neither necessary nor sufficient conditions for an EcoQ and corresponding EcoQO to be useful. In particular circumstances, one or more could be missing from a useful EcoQ – EcoQO, or some additional properties might be considered important. Nonetheless, they were considered excellent properties to use in screening potential EcoQs and EcoQOs. **The more properties from this list that a candidate EcoQ and corresponding EcoQOs lacked, the more likely that the EcoQO would not be a practical and effective guide to actions by managers. If an EcoQO is ineffective at guiding management decision-making, it is not likely to contribute to better protection of marine ecosystems and more sustainable uses of them.**

As reported in Section 5.3.2 of the 2001 WGECO report (ICES, 2001a), metrics (features that are "elements" in the sense of the Bergen Declaration when stated factually, and EcoQOs when stated quantitatively with a reference point) of EcoQs should be:

- relatively easy to understand by non-scientists and those who will decide on their use;

- sensitive to a manageable human activity;
- relatively tightly linked in time to that activity;
- easily and accurately measured, with a low error rate;
- responsive primarily to a human activity, with low responsiveness to other causes of change;
- measurable over a large proportion of the area to which the EcoQ metric is to apply;
- based on an existing body or time series of data to allow a realistic setting of objectives.

In addition an EcoQ metric may:

- Relate to a state of wider environmental conditions.

The background documentation on EcoQOs for the Bergen Declaration (OSPAR, 2002) does not comment on these criteria explicitly. However, commentary on the Purpose and Use of EcoQOs, particularly Parts I.1.7, I.1.9, I.5 and I.6, are consistent with the importance of the criteria adopted by WGECO. Although the criteria were not used in screening the EcoQOs in Table B, Annex 3 of the Bergen Declaration, the text in Appendix I of OSPAR (2002) illustrates that the authors of the Declaration were considering factors similar to those that led to the WGECO criteria, if perhaps in a less systematic way.

3.1.2 Management system needed to implement EcoQOs

Last year's WGECO report (ICES, 2001a) also noted that just having EcoQs and EcoQOs, however well selected, was no assurance of progress towards better protection and more sustainable uses of marine ecosystems. The management system has to use the EcoQOs, and advice provided within an EcoQ – EcoQO framework. WGECO was pessimistic about the prospects for the current management and decision support systems to use the EcoQ – EcoQO framework any more effectively than it used current (and preceding) frameworks for bringing outcomes of fishing into correspondence with goals of protection of ecosystem health and sustainability of uses. To address this problem, in Section 5.1.4.2 (of ICES, 2001a) we noted several necessary attributes of a management system, if it was to use EcoQs and EcoQOs as effective tools in the protection of marine ecosystem health and the achievement of sustainable usage of marine ecosystems:

a) Institutional mechanisms are required to reconcile real or perceived incompatibilities among different objectives, whether they are objectives for fisheries contrasted with integrated objectives for ecosystem quality, or even ecological, economic, and social objectives for any specific use, including (but not exclusively) fishing.

b) For the monitoring and assessment of different EcoQs, it is important to establish a peer review and advisory framework that deals explicitly with quality control of data collection and analysis.

c) For particular metrics used to evaluate an EcoQO, the historic hit, miss, and false alarm rate of the metric should be explicitly examined, and the performance of the metric over time evaluated.

d) It could be argued that it is necessary to protect countless properties of ecosystems, and processes have to be developed to identify which ones should receive direct consideration.

e) Methods need to be developed to ensure that advice is effective in supporting decision-making when progress on achieving numerous individual objectives is uneven.

f) Methods need to be developed to relate specific human activities unambiguously to status relative to specific EcoQOs.

These are relevant considerations in the medium and longer term, as we move from debate about selecting EcoQ elements and EcoQOs to actually using them in management, or even expanding the current pilot tests of Table B, Annex 3 EcoQOs to the whole EcoQ – EcoQO framework, rather than just individual bits of it.

3.1.3 How these criteria are used in this section

The remaining subsections of Section 3 address the EcoQ – EcoQO framework on three time scales: long-term, medium-term, and short-term. These should not be considered an ordering of the sequence in which necessary actions should be inaugurated. Work should be started at the earliest opportunity to make progress on all three time scales. In fact, work relevant to some of the longest term considerations has been on-going for years, inside and outside of WGECO, and must be continued if only with greater urgency. Rather, the three time scales are when the *products* of the activities are needed. The long-term products will be needed only when evaluation of the pilot projects called for in the Bergen Declaration are completed and the scope of the EcoQ initiative is ready to grow from pilot to implementation stages (around 2006–2008). The short-term products are needed almost immediately, for any progress to be made even on the pilots. The medium-term products will be needed in the next 12–18 months (~2003), for additional testing, as in the current pilot projects, to include the EcoQ elements in Annex 3, Table A (Bergen Declaration) that do not yet have corresponding EcoQOs in Table B.

3.2 Longer-term Products Required

The commitments reflected in paragraph 2, and elsewhere in the Bergen Declaration, promise a more inclusive management approach to the North Sea. WGECO particularly welcomes the commitment to immediate action to move ahead on these strategic commitments, as reflected particularly in paragraph 4iii) and associated Annexes 2 and 3. The pilot projects to be run are an appropriate first step to bringing those commitments into practice. However, it must also be acknowledged that the pilots in the Annex are just that: *pilot projects* of how to go about the job of setting EcoQOs for EcoQs, and monitoring status and progress against them. Consistent with the discussion in Section 7 of last year's report (ICES, 2001a) and Section 3.3 here, the selected EcoQOs are generally among the most simple properties that might be informative about EcoQs, and the EcoQs are among the more simple ecosystems properties for which monitoring, evaluation, and reporting can be undertaken. Starting simple is appropriate for a pilot project. However, even if there were to be successful pilots for all the EcoQs in Table A, at that point the EcoQ–EcoQO framework would be very far from "a coherent and integrated set of Ecological Quality Objectives" (Paragraph 4, Bergen Declaration). This, in turn, leaves the science, management, and policy communities far from a framework adequate to implement an ecosystem approach to "manage all human activities that affect the North Sea, in a way that conserves biological diversity and ensures sustainable development" (Paragraph 1, Bergen Declaration).

The Background Document for the Development of EcoQOs for the North Sea (OSPAR, 2002), which was used in the preparation of the Bergen Declaration, also notes that much work remains to be done on EcoQs and EcoQOs. It specfies in its opening text that "There are further objectives relating to a number of issues, where progress has been made, but where further work is required to complete them" (introduction text). Even for EcoQOs which are referred to as "in an advanced stage of development" (paragraph 10), OSPAR points out that the overall work is far from complete. For example, under the EcoQO for Proportion of Oiled Common Guillemots, it is noted that "Oil pollution that affects seabirds comes from a variety of sources … All of these sources of oil will need to be addressed" (Appendix 1, Section 4.1). Under the EcoQO for Commercial Fish Species, it is noted that "In the longer term [it is necessary to] develop biologically and ecologically based target reference levels as a basis for management objectives" (Appendix 1, Section 1.1) (OSPAR, 2002).

WGECO has repeatedly stressed the complexity and scope of the job necessary to have EcoQs sufficient to support the ecosystem approach comprehensively, and how incomplete the scientific ability is at present to support a full suite of EcoQs. At the end of our evaluation of a much larger set of candidate EcoQ elements than is represented in Table A, Annex 3 of the Bergen Declaration, we concluded that "there is a real need for additional development and testing of several metrics described here. There are some components of the ecosystem for which we do not have adequate measures. In the long-term, there is a need for the development of new metrics to describe these key ecosystem qualities" (Section 7.5.4) (ICES, 2001a). Among the ecosystem components for which there were no adequate measures were properties as important as biological diversity, ecological functionality, and spatial integrity. Each of these properties is quite broad, and will require a number of EcoQs to address comprehensively the protection of the entire North Sea and sustainability of all uses of it. Many of the other properties of marine ecosystems that WGECO was able to consider in last year's report are not covered by any of the EcoQs and EcoQ elements in Annex 3, and these must all be brought into the EcoQ – EcoQO framework.

Even properties which are included in Annex 3, and which may appear relatively straightforward, such as "commercial fish species", or a "low level (<2) of imposex in female dogwhelks", require a great deal more work. At present, limit and precautionary biomass reference points for commercially exploited fish stocks have, in almost all cases, been based solely on single-species considerations (ICES, 2001f, 1998) and ignore relationships such as age-size-fecundity (Marshall *et al.*, 1999), spatial and genetic population structure (Section 10, below), and environmental forcing factors (GLOBEC). For an EcoQ – EcoQO-based framework to have a high likelihood of ensuring protection of even commercially exploited species as components of marine ecosystems with important structural and functional roles, all these considerations will eventually have to be addressed. Imposex level in dogwhelks is a good measure of TBT in local areas, but TBT is only one of many contaminants that impact ecosystem health, and its use will soon be banned in the North Sea (C. Frid, pers. comm.). Other EcoQ elements and EcoQOs are necessary for the effects of contaminants to be captured in the EcoQ – EcoQO framework.

WGECO, ICES, and the larger community will be monitoring progress on the pilot projects for at least two reasons. The first reason is as a test of the effectiveness of the EcoQ – EcoQO framework as a measurement and evaluation tool. Can technical experts (scientists and their co-workers) actually identify and justify reference levels, measure status and trends relative to the reference points, and provide sound risk-based advice using EcoQOs? Does the EcoQ – EcoQO framework clearly inform all stakeholders of the questions posed and answers found by the technical experts, and more generally, of the state of the ecosystem relative to their interests and values? Finally, does the EcoQ – EcoQO framework provide clear guidance to managers and decision-makers relative to the decisions for which they must take responsibility, the options available to them for each decision, and the consequences of each option for ecosystem conservation and sustainability of activities. It is expected that the effectiveness of the EcoQ – EcoQO framework will evolve over time, as, for example, the framework of providing fisheries advice within a precautionary approach has evolved within ICES (ICES PAWG report 1997, 2001). However, the pilot projects should provide some indication of at least whether the EcoQ – EcoQO framework is a promising pathway to improved advice, improved decision-making, and improved public understanding.

The second reason to monitor the progress of these pilot projects closely is as a test of the commitments of governments and agencies to use the EcoQs as a basis for management action. Even if the EcoQ – EcoQO framework is an effective tool, to be worthwhile the tool must be used for improving decision-making about the uses of marine ecosystems, including, but not restricted to, fishing. This cannot be a formal test, of course, because there are numerous reasons why marine ecosystems (and the Ecological Quality elements captured in the EcoQOs) may change, and not all of them may be the result of decision-making based on the EcoQ – EcoQO framework. Nonetheless, the pilot projects have to be informative about the role of the new EcoQ – EcoQO framework, and its constituent tools, in decision-making. If it appears that the tool is not used, one must ask what justification there is to invest the large amount of scientific effort that will be required to make the EcoQ – EcoQO framework adequately comprehensive. Remembering that the pilot projects soon to be under way from Table B, Annex 3 are among the simplest cases, if the EcoQ – EcoQO framework is going to result in noticeable improvements in the ecosystem properties targeted by EcoQ elements, it should do so in these simple cases. If not, the EcoQ – EcoQO framework can hardly be expected to lead to major gains in ecosystem protection and sustainability of uses when the management tasks are made much more complex: more EcoQOs to satisfy simultaneously, demanding action on the basis of ecosystem properties less convincingly linked to direct benefits, etc.

IF the EcoQ – EcoQO framework seems to be effective, and IF the institutional buy-in is strong enough that progress seems to be being made, the scientific community is expected to be active in developing the more comprehensive suite of EcoQs and EcoQOs that we believe necessary. From the reports produced over the past twelve years, WGECO feels that it has demonstrated that it has important contributions to make to these scientific tasks, adding the necessary complexity and completeness, while keeping the whole suite of EcoQs and EcoQOs operational and realistic.

Recommendation from WGECO to ACE and MCAP: WGECO should be identified as the lead Working Group to coordinate the scientific input needed for advisory support to the developing EcoQ – EcoQO framework. Some of this coordination will be through work done by WGECO, and some through evaluation against consistent standards and integration of work done in other Working and Study Groups.

3.3 Short-term Considerations

The Ministers signing the Bergen Declaration in March 2002 have adopted ten EcoQOs (Table 3.3.1) for immediate application as a North Sea pilot project within the framework of OSPAR. The pilot project will:

a) assess the information that is, or can be made, available in order to establish whether the EcoQOs are being, or will be, met. Where the EcoQOs are not being met, the information will be used to determine the reason. Costs and practicability should be taken into account in deciding what information can be made available;

b) where an EcoQO is not being met, review any policies and practices which are contributing to that failure; and

c) if needs be, reconsider the formulation of such EcoQOs (Paragraph 4iv, Bergen Declaration).

Plainly, in order to assess whether or not an EcoQO is being met, there is a need to establish monitoring schemes. The Ministers thus agreed to establish coherent monitoring arrangements, in order to enable progress towards meeting the EcoQOs to be assessed. These arrangements will be integrated into the OSPAR Joint Assessment and Monitoring Programme (Bergen Declaration, Paragraph 4v).

Table 3.3.1. Ecological quality elements and objectives agreed by North Sea Ministers at Bergen, March 2002 (from Annex 3, Table B, Bergen Declaration). Further background information on each element/objective may be found in OSPAR (2002).

Ecological quality element	Ecological quality objective

(a) Spawning stock biomass of commercial fish species	Above precautionary reference points[1] for commercial fish species where these have been agreed by the competent authority for fisheries management
(c) Seal population trends in the North Sea	No decline in population size or pup production of ≥10 % over a period of up to 10 years
(e) By-catch of harbour porpoises	Annual by-catch levels should be reduced to levels below 1.7 % of the best population estimate
(f) Proportion of oiled common guillemots among those found dead or dying on beaches	The proportion of such birds should be 10 % or less of the total found dead or dying, in all areas of the North Sea
(m) Changes/kills in zoobenthos in relation to eutrophication[2]	There should be no kills in benthic animal species as a result of oxygen deficiency and/or toxic phytoplankton species.
(n) Imposex in dogwhelks (*Nucella lapillus*)	A low (<2) level of imposex in female dogwhelks, as measured by the *Vas Deferens* Sequence Index
(q) Phytoplankton chlorophyll *a*[2]	Maximum and mean chlorophyll *a* concentrations during the growing season should remain below elevated levels, defined as concentrations > 50 % above the spatial (offshore) and/or historical background concentration
(r) Phytoplankton indicator species for eutrophication[2]	Region/area-specific phytoplankton eutrophication indicator species should remain below respective nuisance and/or toxic elevated levels (and increased duration)
(t) Winter nutrient concentrations (Dissolved inorganic nitrogen (DIN) and dissolved inorganic phosphate (DIP)) [2]	Winter DIN and/or DIP should remain below elevated levels, defined as concentrations > 50 % above salinity-related and/or region-specific natural background concentrations
(u) Oxygen[2]	Oxygen concentration, decreased as an indirect effect of nutrient enrichment, should remain above region-specific oxygen deficiency levels, ranging from 4–6 mg oxygen per litre

Notes:

[1]In this context, "reference points" are those for the spawning stock biomass, also taking into account fishing mortality, used in advice given by ICES in relation to fisheries management.

[2]The ecological quality objectives for elements (m), (q), (r), (t) and (u) are an integrated set and cannot be considered in isolation. ICES will give its further advice during the implementation phase.

Many of these EcoQOs either correspond to, or address ecological features similar to, potential EcoQs discussed in Section 7 of last year's report (ICES, 2001a). In some cases, concerns raised last year remain relevant to the EcoQOs that have been selected for the pilot project, and in some other cases the EcoQO selection raises new concerns relative to our criteria (Box 3.1).

Box 3.1. Detailed comments on individual EcoQOs from the Bergen Declaration, Annex 3, Table B.

Spawning stock biomass of commercial fish species

Precautionary fishing mortality rates are implicitly included in this objective according to the footnote in the Bergen Declaration. The use of the word "above" is confusing and might better be replaced by "beyond". The description in the background document only refers to biomass reference points and the text should therefore be amended.

Seal population trends in the North Sea

The background document does not specify whether the objective applies to the total North Sea stocks of grey seals and common seals, respectively, or to largely reproductively isolated sub-populations. This needs to be resolved. It is by no means clear how the objective must be interpreted: a decline > 10 % within a single year would obviously not meet the objective, even when it is followed by an increase. Moreover, the objective would allow for, e.g., a 40 % decrease over a 50-year period, if only the condition is met that the decline is so gradual that it does not exceed 10 % within 10 years.

By-catch of harbour porpoises

There is a potential statistical problem, because the objective does not state the probability for any estimate being below 1.7 %. The North Sea harbour porpoise population may have some sub-divisions; further research is under way to resolve this issue. The current EcoQO for the North Sea assumes a unit stock; adjustments would be required in the light of emerging research results.

Proportion of oiled common guillemots among those found dead or dying on beaches

The background document clearly states that this objective does not refer to specific localities or events, but to monitoring records integrated over areas and time. It is by no means clear which areas are distinguished or whether the temporal unit is season or year.

Changes/kills in zoobenthos in relation to eutrophication

This objective might put unrealistic demands on monitoring efforts, unless some kind of warning system could be developed to trigger extensive survey activities.

Imposex in dogwhelks (*Nucella lapillus*)

This is also a completely open-ended objective that might require sampling at every location where dogwhelks might occur.

The Ministers noted that ICES should collaborate with OSPAR to review progress on the pilot projects testing the EcoQOs in Table 3.3.1 (Paragraph 4vi), but gave no details for roles and responsibilities. ICES can offer scientific advice and input at several stages of the short-term EcoQO process. First, as explained last year (Section 5.3 of ICES, 2001a) all of the adopted EcoQOs for the pilot project should be evaluated against the criteria outlined in Section 3.1.1, above. Some of the EcoQOs were considered in 2001 by WGECO (ICES, 2001a), and these considerations are repeated below, with some further development. We continue to recommend that the other EcoQOs undergo the same evaluation, and note that other Working Groups should provide their input to this process.

Secondly, ICES, through the expertise of its Working Group and advisory process, could have a role in coordinating the monitoring required for many of the EcoQOs and/or in evaluating the results of this monitoring. Some input would have broad and general value, such as the advice on trend monitoring in Annex 9 of the 2001 ACME Report (ICES, 2001e). There are also many more specific opportunities for ICES involvement in the work associated with individual EcoQOs, as given below.

3.3.1 Evaluation of EcoQOs

In 2001, WGECO graded the various possible metrics for properties covering key ecological qualities (ICES, 2001a). Table 3.3.1.1 repeats this analysis for the ten EcoQOs selected in Annex 3, Table B of the Bergen Declaration.

3.3.2 Possibilities to improve the performance of the EcoQ metric

3.3.2.1 Commercial fish species

Measurement of spawning stock biomass is subject to a number of errors and biases that are well known to relevant ICES working groups and to advisory committees. ICES is continually striving to improve the situation in order to provide better fish stock advice; these improvements will help improve the performance of the EcoQ metric.

Fish stocks will always be responsive to natural factors that cannot be controlled. Understanding of the effects of these natural factors will improve through time, but full understanding is unlikely to be achieved in the near future, if ever.

3.3.2.2 Seal population trends

It is not known how sensitive seal populations are to human activities apart from direct killing, either deliberately or through fisheries by-catch (where the linkage can be modelled and is relatively tight). However, this EcoQO is designed to act as a trigger for further research to determine whether manageable human activities are the cause of any future decline. In the meantime, research on aspects of the interaction between humans and seals will continue, for instance, on establishing cause-effect relationships between pollutants and seal population health. The greatest recent cause of negative change in seal populations was due to an epizootic; the degree to which this was an indirect result of chemical pollution is the subject of debate and research.

3.3.2.3 By-catch of harbour porpoises

Harbour porpoise by-catch is not easy to measure accurately, as it requires the deployment of independent observers on reasonable proportions of the fleets causing the by-catch. In general, recommended methods are being used in the two existing schemes (UK and Denmark) that are examining harbour porpoise by-catches in the North Sea. Norwegian by-catches have never been monitored using reliable methods. In addition, monitoring of small-boat fisheries is problematic everywhere, and the results are less comprehensive than for the fisheries using larger boats. In all cases, greater monitoring effort is required if annual figures are to be used. This is also required for EU member states under the Habitats Directive.

3.3.3 Development of the scientific role of ICES in relation to the pilot project on EcoQOs

WGECO has expertise on the matters addressed by the selected EcoQ metrics. Broadly, once an EcoQ metric has been decided, science can help in defining the current level of that metric, reconstructing the historical trajectory of that metric, and in establishing and conducting a scientifically robust monitoring programme. Monitoring information, or other research information, might be used to determine what management actions could be taken to help meet the EcoQO, particularly when placed in the context of historical values of the metric. WGECO has applied its expertise to some of the pilot project EcoQOs to illustrate the role that ICES might play in future.

Obviously, WGECO is not the only Working Group within ICES with expertise in the scientific disciplines relevant to the EcoQOs in Table 3.3.1. Input should also be encouraged from other appropriate Working and Study Groups. However, we encourage that the evaluation framework tested and adopted by WGECO be used widely, and that the role of ICES in the EcoQ initiative continue to build on the experience of WGECO in provision of the scientific basis for ICES advice on ecosystem impacts of human activities. For example, the experts in the Working Group on Seabird Ecology (WGSE) and the Working Group on Marine Mammal Population Dynamics and Habitats (WGMMPH) (see Sections 3.3.2.2–3.3.2.4, below) are generally not from institutes presently supported by governmental funds for participation in ICES. Thus, although their expertise is very relevant, there are relatively fewer guarantees of participation in ICES meetings by these experts. ICES and its North Sea Member Countries might need to address this issue if they wish to attract EcoQO-related work in future.

3.3.3.1 Spawning stock biomass of commercial fish species

ICES is currently the source of scientific advice on the current and historical Spawning Stock Biomasses (SSB) for commercially exploited species in the North Sea. Notwithstanding the criticisms that have been directed at ICES advice on SBB (and fishing mortality) as inaccurate and imprecise (letter from the EU addressed by MCAP), ICES advice has been a reliable basis for management decision-making (see ICES Strategic Plan, ICES website). ICES has also introduced a number of quality assurance steps to its methods for estimating stock status, including SSB, and sources of error and bias, when they occur, are generally understood (ICES, 2002a, 2002b). ICES has also considered the value of B_{lim} and B_{pa} as EcoQO reference levels, and found them to be appropriate (ICES, 2001b). As long as the ICES intent that management decision-making keep stocks above B_{pa} with high probability is achieved, such an approach is consistent with the intent of using EcoQs and EcoQOs to maintain healthy marine ecosystems.

WGECO recommends that ICES continue to be the source for scientific advice on current and historical SSB for exploited fish stocks, and supports strongly the implementation of further quality control measures, as they are identified. ICES has also advised routinely on management measures to increase SSB when necessary. WGECO interprets the commitments in the Bergen Declaration (Section III – Sustainable Fisheries) as a mandate to increase the scope and clarity of such recommendations, with ACFM receiving input from groups such as WGECO, as well as the assessment Working Groups, when developing its advice.

3.3.3.2 Seal population trends in the North Sea

WGMMPH and its predecessors have periodically assessed seal populations of the North Sea and are the only existing international group in a position to do this. Therefore, WGECO recommends that ICES be tasked to lead the scientific implementation of this EcoQO, collating, evaluating, and integrating the census efforts of the various countries around the North Sea. The majority of grey seals in the North Sea haul out on UK coasts and are monitored by an annual programme, using standardised methods, conducted by the Sea Mammal Research Unit. Harbour seals occur in approximately equal numbers on continental coasts and UK coasts. They are not monitored annually on UK coasts, but are monitored elsewhere. Methods are standardised. Less is known about the causes of changes in seal populations, but some projects suggested and fostered by WGMMPH will help in determining the effects of contaminants on seal populations. The Working Group could also suggest and foster projects to examine other factors that might contribute to changes in seal populations. All monitoring must obviously reflect the defined stock structure. As specific tasks, ICES could publish a) a standardised seal censusing manual, and b) an annual report on the state of North Sea seal populations.

3.3.3.3 By-catch of harbour porpoises

WGMMPH and its predecessors have reviewed small cetacean by-catch on two occasions, most recently in 2001 (ICES, 2001b). Again, WGECO recommends that ICES be tasked to lead the scientific implementation of this EcoQO, collating, evaluating, and integrating the census efforts of the various countries around the North Sea. In 1998 WGMMPH also reviewed methods for monitoring such by-catch, and recommended protocols for producing reliable results. Failure of countries to allocate greater effort to monitoring harbour porpoise by-catch will mean that the status of, and progress with, this EcoQO will be impossible to evaluate reliably.

The metric also requires assessment of by-catch against an overall population figure; the figure currently in use derives from surveys in 1994, so there is a need to update this in the near future. Plans to repeat the survey in 2003 or 2004 are being made at present. Methods are reasonably standardised by the IWC Scientific Committee and WGECO recommends that the protocols recommended by the IWC Scientific Committee be followed.

Management action to reduce by-catch could be through: a) time/area closures, b) modification of fishing gear, or c) overall effort reduction. All of these actions require knowledge of fishing fleets, including the number of vessels, fishing methods, fishing areas, and measures of fishing effort. ICES requires such information on effort for many advisory roles that it undertakes, but WGECO and other groups have failed consistently to obtain this information. Advice on the appropriateness and effectiveness of options for mitigation measures has been sought of, and provided by, ICES in recent years, and we do not consider it a priority to revisit this advice in the near future.

Table 3.3.1.1 Evaluation of EcoQ metrics by WGECO. Metrics were graded against those features considered to be qualities of good EcoQOs (ICES, 2001a). Dark-shaded rectangles fully match the criterion; lightly-shaded rectangles do not fully match the criterion and further improvements (where considered possible) are discussed in the section indicated. Because the primary expertise for eutrophication, phytoplankton, nutrients, and oxygen within ICES is present in other Working Groups, WGECO did not complete five rows of this table. However, we encourage the appropriate experts within ICES to complete this table, with accompanying text where needed, so the entire table can be brought forward in the ICES advice.

Ecological quality element	a) Sensitive	b) Linked	c) Low error	d) Responsive	e) Measurable	f) Time series
Spawning stock biomass of commercial fish species			3.3.2.1	3.3.2.1		
Seal population trends in the North Sea	3.3.2.2	3.3.3.2		3.3.3.2		
By-catch of harbour porpoises			3.3.3.3			
Proportion of oiled common guillemots among those found dead or dying on beaches						
Changes/kills in zoobenthos in relation to eutrophication						
Imposex in dogwhelks (*Nucella lapillus*)						
Phytoplankton chlorophyll *a*						
Phytoplankton indicator species for eutrophication						
Winter nutrient concentrations						
Oxygen						

Notes:

a) Sensitive to a manageable human activity.

b) Relatively tightly linked in time to that activity.

c) Easily and accurately measured, with a low error rate.

d) Responsive primarily to a human activity, with low responsiveness to other causes of change.

e) Measurable over a large proportion of the area to which the EcoQ metric is to apply.

f) Based on an existing body or time series of data to allow a realistic setting of objectives

3.3.3.4 Proportion of oiled common guillemots among those found dead or dying on beaches

Standards for conducting beached birds surveys have been established by OSPAR (OSPAR, 1995), and WGECO endorses those protocols. Currently only one international survey occurs each year (in February), with surveys at other times being more systematic in some countries than in others. Monitoring is already included in the Trilateral Monitoring and Assessment Programme (TMAP) in the Wadden Sea. Nevertheless, surveys around the North Sea could be more frequent and better coordinated, perhaps on a monthly basis. It would be appropriate for WGSE to review the sampling structure needed to ensure that reliable basin-wide information is available for the North Sea. WGSE could also collate results, establish trends, and report on status relative to historical rates of oiling.

3.3.3.5 Changes/kills in zoobenthos in relation to eutrophication

The Marine Chemistry Working Group (MCWG) considered this issue at its March 2002 meeting (Text Box 3.3), and further examination will occur at the Benthos Ecology Working Group (BEWG) meeting in April 2002.

3.3.3.6 Imposex in dogwhelks (*Nucella lapillus*)

This EcoQO will provide a basis for the assessment of the recovery of the marine ecosystem following implementation of the IMO restrictions on TBT from 2003 and the outright ban from 2008.

WGECO recommends that the Working Group on Biological Effects of Contaminants (WGBEC) or related working groups examine the science base and monitoring requirements for this EcoQO.

3.3.3.7 Phytoplankton chlorophyll *a*

MCWG considered this issue at its March 2002 meeting (Text Box 3.3); WGECO recommends that the Working Group on Phytoplankton Ecology (WGPE) or related working groups examine the science needs for this EcoQO.

3.3.3.8 Phytoplankton indicator species for eutrophication

MCWG considered this issue at its March 2002 meeting (Text Box 3.3). WGECO recommends that the ICES/IOC Working Group on Harmful Algal Bloom Dynamics (WGHABD) or related working groups examine the science needs for this EcoQO.

3.3.3.9 Winter nutrient concentrations (Dissolved inorganic nitrogen (DIN) and dissolved inorganic phosphate (DIP))

MCWG considered this issue at its March 2002 meeting (Text Box 3.3).

3.3.3.10 Oxygen

MCWG considered this issue at its March 2002 meeting (Text Box 3.3).

Text Box 3.3 Relevant text from the March 2002 meeting of the ICES Marine Chemistry Working Group.

"*The discussion in the subgroup was mainly based on the paper 'Current status of Ecological Quality Objectives for the Greater North Sea with regard to Nutrients and Eutrophication Effects' (EcoQOs-eutro, EUC 01/5/3 – Rev 1).*

The group agreed in principle that it was necessary to have objective criteria for assessing nutrient enrichment and ecological quality objectives. However, some of the 'Agreed Harmonised Assessment Criteria' require clarification and may not be relevant to all sites at all times.

The OSPAR region—in common with other coastal areas—is subject to large natural temporal and spatial variations in nutrient concentrations. One of the major deficiencies of the proposed criteria is that transboundary nutrient transports are not adequately taken into account. This is particularly important for inorganic nutrients since the natural fluxes in the North Sea are many orders of magnitude greater than the anthropogenic fluxes, which are also likely to be localised in space and time. Care must therefore be used in interpreting nutrient data, since misleading or inappropriate conclusions may be drawn. For example, 'winter concentrations' of nutrients are only appropriate for the description of phytoplankton development in summer if it is confirmed that transboundary effects are not significant over the intervening period. Moreover, the definition of 'winter concentrations' is too broad as this parameter is defined by the status of the ecosystem (maximum accumulation of nutrients and minimum primary productivity) and not by a specific time of the year. The start of the phytoplankton spring bloom may not necessarily occur at the same time for all stations.

We are also concerned that the criteria listed as 'Assessment Criteria' are not necessarily universally applicable and recommend that the listed criteria be checked for relevance to local conditions. For example, natural perturbations such as wind-induced mixing or upwelling need to be considered before deciding whether critical values have been exceeded. The rationale for assigning values to 'background concentrations' and 'elevated concentrations' is not always clear since we have limited information as to how the 'spatial/historical background concentrations' were fixed. The relevant information needs to be readily available. In addition, if the normal concentration of a nutrient is low, an increase of >50 % may not be environmentally significant. It is also not clear from the document what criteria will be used to define the boundaries of problem areas (PA). Given these concerns, it is surprising that, in Document EUC 01/5/2-Add.1-E, Item 9), no comments are included under the heading 'Remaining problems and suggested actions'.

The scientific background behind the Ecological Quality Objectives, strategies to support their evaluation and information on their proper use needs more clarification and ongoing discussion"

References

Anon. 1999. Workshop on Ecological Quality Objectives (EcoQOs) for the North Sea. 1–3 September 1999, Scheveningen, the Netherlands. Nordic Council of Ministers. TemaNord 1999:591. 75 pp.

ICES. 1997. Report of the Study Group on the Precautionary Approach to Fisheries Management. ICES CM 1997/Assess:7.

ICES. 1998. Report of the Study Group on the Precautionary Approach to Fisheries Management. ICES CM 1998/ACFM:10

ICES. 2001a. Report of the Workng Group on Ecosystem Effects of Fishing Activities. ICES CM 2001/ACME:09.

ICES. 2001b. Report of the ICES Advisory Committee on Ecosystems, 2001. ICES Cooperative Research Report, 249.

ICES. 2001c. Report of the ICES Advisory Committee on Fishery Management, 2000. ICES Cooperative Research Report, 242: 1–2.

ICES. 2001d. Report of the Working Group on Seabird Ecology. ICES CM 2001/C:05.

ICES. 2001e. Report of the ICES Advisory Committee on the Marine Environment, 2001. ICES Cooperative Research Report, 248: 173–179.

ICES. 2001f. Report of the Study Group on Further Development of the Precautionary Approach to Fisheries Management. ICES CM 2001/ACFM:11.

ICES. 2002a. Report of the Study Group on the ACFM Working Procedures. ICES CM 2002/MCAP:01.

ICES. 2002b. Report of the Meeting of the Management Committee on the Advisory Process, January 21–23, 2002. ICES CM 2002/A:09.

Kabuta, S., and Enserinck, L. 2000. Development of indicators for the Dutch North Sea – project GONZ. ICES CM 2000/ACME: WP7.

Lanters, R.L.P., Skjoldal, H.R., and Noji, T.T. (eds.) 1999. Ecological Quality Objectives for the North Sea. Basic document for the Workshop on Ecological Quality Objectives for the North Sea, 1–3 September 1999, Scheveningen, The Netherlands. RIKZ Report 99.015. Fisken og Havet, 10–1999.

Marshall, C.T., Yaragina, C.S., Lambert, Y., and Kjesbu, O.S. 1999. Total lipid energy as a proxy for total egg production by fish stocks. Nature, 402: 288–290.

OSPAR. 1995. JAMP guidelines on standard methodology for the use of beached birds as indicators of marine oil pollution. OSPAR Commission Monitoring Guidelines, 1995–6.

OSPAR. 2002. Background document on the development (within the OSPAR framework) of Ecological Quality Objectives (EcoQOs) for the North Sea. Paper to the meeting of the Committee of North Sea Senior Officials, Oslo, 9–10 January 2002. DOC CONSSO JAN 02/3/24.

Piet, G.J. 2001. Development of ecological quality objectives for the North Sea fish community. Working paper at the 2001 WGECO meeting.

3.4 The Medium Term

For several EcoQ elements in Table A of Annex 3 of the Bergen Declaration, EcoQOs have not been proposed in Table B. For such elements, in cases where there is presently adequate knowledge for setting EcoQOs, the following sub-sections provide guidance to the relevant scientific information, much of which has already been provided by ICES. In those cases, the impediment for progressing to an EcoQO seems to be decisions by managers and policy-makers not to set reference levels for the ecosystem metrics. As discussed in last year's report in Section 5.3.1, it is for society, informed by science, to make those choices. Where appropriate, this section offers suggestions for how science can inform the discussions about the policy decision more fully.

For several elements where the current scientific knowledge is either inadequate, or possibly adequate but not properly consolidated, the sub-sections lay out as specifically as possible the steps that must be pursued to gain or consolidate the necessary scientific basis for setting EcoQOs. In some cases, those science programmes will be demanding, but they are necessary before it will be appropriate to complete the identification of specific EcoQOs.

3.4.1 Issue 2 – Threatened and declining species

The current EcoQ element in Table A states:

(b) Presence and extent of threatened and declining species in the North Sea.

3.4.1.1 Convergence of threatened and declining species listing and EcoQO development

The general formulation of this EcoQ element implies that considerable work needs to be done to develop operational EcoQOs for this issue. The starting point for this work is the listing of threatened and declining species on the basis of the Texel/Faial criteria by the OSPAR Biodiversity Committee (BDC) (OSPAR, 2000). This Committee has scheduled discussion on the final version of these criteria in the autumn of 2002. The draft criteria and guidance on how to interpret these criteria are given in Tables 3.4.1.1.a and 3.4.1.1.b. The next step to be made within OSPAR will be to apply these criteria in order to select species and habitats that need to be protected.

Table 3.4.1.1.a. Draft Texel/Faial criteria: selection criteria for species to be listed as threatening and declining.

	Global importance: Global importance of the OSPAR Area for a species. Importance on a global scale, of the OSPAR Area, for the species is when a high proportion of a species at any time of the life cycle occurs in the OSPAR Area.
	Local importance: Importance within the OSPAR Area, of the regions for the species where a high proportion of the total population of a species within the OSPAR Area for any part of its life cycle is restricted to a small number of locations in the OSPAR Area.
	Rarity: A species is rare if the total population size is small. In case of a species that is sessile or of restricted mobility at any time of its life cycle, a species is rare if it occurs in a limited number of locations in the OSPAR Area, and in relatively low numbers. In case of a highly mobile species, the total population size will determine rarity.
	Sensitivity: A "very sensitive" species is one if very easily adversely affected by a human activity, and/or if affected is expected to only recover over a very long period, or not at all. A "sensitive" species is one if easily adversely affected by a human activity, and/or if affected is expected to recover in a long period.
	Keystone species: a species which has a controlling influence on a community.
	Decline: means an observed or indicated significant decline in numbers, extent or quality (quality refers to life history parameters). The decline may be historic, recent or current. "Significant" need not be in a statistical sense.

Table 3.4.1.1.b. Guidance on the selection criteria for species.

<table>
<tr><th>Criterion</th><th>Guidance</th></tr>
<tr><td>1</td><td>“High proportion” is considered to be more than 75 %, when known.</td></tr>
<tr><td>2</td><td>“High proportion” is considered to be 90 % of the population in a small number of locations of 50 km × 50 km grid squares. This is dependent on scientific judgement regarding natural abundance, range or extent and adequacy of recording. A different scale may be needed for different taxa.</td></tr>
<tr><td>3</td><td>“A limited number of locations”could be in a small number of 50 km × 50 km grid squares, but a different scale may be needed for different taxa. This is dependent on scientific judgement regarding natural abundance, range or extent and adequacy of recording. Species which are present in high abundance outside of the OSPAR Area and only occur at the edges of the OSPAR Area will not generally qualify as “rare” species.</td></tr>
<tr><td>4</td><td>A “very long period” may be considered to be more than 25 years and “long period” in the range of 5 to 25 years. The time frame should be on an appropriate scale for that species.
Sensitivity to human activities is measured by:
a. life history characteristics;
b. dependence on other specific ecological attributes, e.g., restricted/specific habitat requirements.</td></tr>
<tr><td>5</td><td>No guidance</td></tr>
<tr><td>6</td><td>“Decline” is divided into the following categories:
a) Extirpated (extinct within the OSPAR Area): a population of a species formerly occurring in the maritime area is defined as extirpated:
• if it was still occurring in the area at any time during the last 100 years;
• and if there is a high probability, or it has been proved, that the last individuals have since died or moved away;
• or if surveys in the area have repeatedly failed to record a living individual in its former range and/or known or expected habitats at appropriate times (taking into account diurnal, seasonal, annual patterns of behaviour) for at least ten years.
b) Severely declined: a population of species occurring in the maritime area is defined as severely declined:
• if individual numbers show an extremely high and rapid decline in the area over an appropriate time frame, or the species has already disappeared from the major part of its former range in the area;
• or if individual numbers are at a severely low level due to a long, continuous and distinct general decline in the past.
c) Significantly declined: means a considerable decline in number, extent or quality beyond the natural variability and in an appropriate time frame for that species.
d) High probability of a significant decline in number, extent or quality in the future.</td></tr>
</table>

WGECO foresees several difficulties when applying these criteria (cf. Text Box 3.4.1).

Box 3.4.1. Selection of threatened and declining species – lessons learned.

There has been substantial debate about the appropriate criteria for evaluating marine species, particularly ones exploited commercially, with regard to risk of extinction (or regional extirpations). The quantitative criteria for listing species at various categories of risk, developed by the Species Specialist Committee of the International Union for Conservation of Nature (Mace *et al*., 1996), have been adopted, with minor variants, by the IUCN, CITES, and several countries. Reviews by fisheries experts have concluded that the criteria pose problems when applied to marine species (FAO, 1999; Powles *et al.*, 2000). The criteria for absolute population numbers and absolute range may be too liberal. Marine species that are at some risk of disappearance may not meet the empirical standards, or may be impossible to sample with sufficient accuracy to evaluate on the criteria. In contrast, the decline criterion (50 % decrease in abundance in the longer of ten years or three generations) is widely considered to be too conservative. Many marine species show such fluctuations without risk of extinction or extirpation (FAO, 1999). Although the text of the IUCN rules note that "natural fluctuations" should not be grounds for listing, the burden of proof requires that there be clear evidence that the fluctuation is natural, which is rarely possible with a species exploited or taken commonly as by-catch. A special IUCN Working Group (IUCN, 2000, 2001), a team of U.S. scientists (Musick, 1998, 1999), FAO (FAO, 1999), and Hutchings (2001) all reviewed these arguments, and came to different conclusions in each case. However, all the reviews agreed on the need for clear quantitative guidelines (not rigid rules), to make the listing process as objective and consistent as possible. The qualitative approach of the OSPAR BDC avoids some of the debates about the correct overall values for maximum tolerable decline, minimum population size, etc. However, it does not replace them with other objective, empirical guidelines that will make consistent application across species easier in practice. WGECO expects that similar debates will occur when the Texel/Faial criteria are applied on a case-by-case basis, and differences among species must be accommodated by qualitative and, at best, semi-quantitative guidelines.

Notwithstanding possible difficulties in application of the Texel/Faial criteria, and opportunities that we may see for improvements to the criteria and listing process, this Term of Reference requires that WGECO addresses how to develop EcoQOs for species that the OSPAR selection process lists as threatened or declining. WGECO will proceed with that task, working the output of the OSPAR selection process. WGECO considers what properties a species listed as threatened and declining species should have, in order to develop robust and effective EcoQOs. We base our consideration on several of the properties identified in Section 3.1.1, above, as characteristics of good EcoQOs. Where EcoQOs are set for threatened and declining species that perform poorly in the evaluation we outline below, WGECO expects that difficulties may occur in implementation, monitoring, and/or evaluation of progress on the EcoQO.

3.4.1.2 The way ahead

WGECO expects that not all species listed as threatened and declining by OSPAR will be suitable for use in setting robust and effective EcoQOs. There are likely to be species on the OSPAR list whose metrics of status do not meet one or several of the WGECO criteria for good EcoQOs. In at least some cases, the areas of failure may be important considerations in setting operational EcoQOs, such as ability to measure and responsiveness to human activities. WGECO proposes a series of steps to be followed to determine which species on the OSPAR list would be suitable for robust and effective EcoQOs, applying a subset of our criteria particularly relevant to the feasibility of the pilot project. We stress that this treatment should not be taken as implying that threatened and declining species that are not best suited for EcoQOs do not need conservation action. Rather, the protection and restoration of threatened and declining species which are not suitable for setting effective EcoQOs would better be achieved in ways other than via the EcoQ – EcoQO framework.

<u>Step 1 – Establish whether the species occurs in the Greater North Sea (OSPAR Region II).</u>

This is the area covered by the Bergen Declaration. Species which are vagrants or which do not occur in the Greater North Sea should not be selected for setting EcoQOs, at least under the provisions of the Bergen Declaration.

<u>Step 2 – Establish whether the status of the species can be quantified accurately.</u>

This step applies to our criterion that effective EcoQOs can be **easily and accurately measured, with a low error rate**. If the status of the species cannot be quantified accurately or precisely, it is not appropriate to set a quantitative EcoQO for the species, and it would be very difficult to monitor status relative to the EcoQO.

Note: We expect some of the rarest species, of potentially greatest concern, to have abundances that cannot be measured accurately, just because they will be rarely encountered in surveys. Priority should be given to developing properties of these species that can be monitored reliably, so information is available regularly on the success of efforts to conserve and recover these species at possibly greatest risk.

Step 3 – Establish why the species is threatened or declining.

This step applies our criteria of **sensitive to a manageable human activity, and responsive primarily to a human activity, with low responsiveness to other causes of change.**

If the main causes of the decline can be established, and factors under management control play a strong enough role in the decline that it is realistic to expect population responses to management actions, then it is possible to proceed further. If the main causes are not related to manageable human activities, the species is not suitable for setting EcoQOs.

Note: If the causes of a species being listed as threatened or declining are not primarily human activities, it still may be necessary for conservation measures to be implemented. Some of these might affect human activities, even if they are not the major threats. These conservation and recovery efforts would better be undertaken outside an EcoQ – EcoQO framework. If the causes of a decline are unclear, more scientific study would be needed urgently. It would be inappropriate, though, to set EcoQOs for such species before the studies clarified the contribution of human activities to the declines.

Step 4 – Establish whether trends in population status can be detected reliably on time frames relevant to management (perhaps over 5 years).

This step applies to our criteria that an EcoQO should **be tightly linked in time to the human activity affecting the trend**, and to management actions to modify the activity. The criterion that an EcoQO should be **easily and accurately measured** is also relevant to this step. It should be possible to detect trends in population status reliably on time frames relevant to management (perhaps over five years). Regular monitoring and evaluation would provide feedback on the effectiveness of the management measures at improving "environmental health" of the sea. If yes, then it is possible to proceed with setting robust EcoQOs for the species. These would probably be associated with an abundance, range or other property that would be taken as a secure status for the species. The values, and the process leading to selecting them, would be species specific. However, simulation modelling should be an important tool in setting such EcoQOs (Burgman *et al*., 1992), unless there is a **long time series of reliable data on population status** (another of our criteria), including a time when the population was considered secure. Then regular monitoring and evaluation would provide feedback on progress towards the EcoQO, and simultaneously on effectiveness of the management measures at improving "environmental health" of the sea.

If it is not possible to detect trends in the indicator(s) of population status over reasonable time frames, then it is not possible to evaluate status relative to an EcoQO, or the effectiveness of management. In such cases, the species is not well-suited for use in the EcoQ – EcoQO framework. Again protection and restoration measures might still be important, but they would have to be implemented with the knowledge that feedback on their effectiveness would be available only on very long time scales.

Gubbay (2001) suggested nine classes of factors that make a species especially sensitive to decline. These include for instance:

- species that are very large, long-lived and/or have low fecundity;
- species that are or have been subject to over-exploitation;
- species that are subject to large-scale mass mortality.

In order to make a particularly informative set of EcoQOs for threatened and declining species, if enough species were evaluated positively on our four-step process, pilot EcoQOs could be chosen such that a broad range of these classes were covered. It is possible, though, that most or all threatened and declining species that are large, long-lived, and have low fecundity will be so rare that they score poorly at step 2, whereas many species subject to mass mortalities might score poorly on step 3 or 4. If that result is found, it may be more effective to seek frameworks other than the EcoQ –EcoQO framework in which to undertake protection and restoration of such species.

Clearly many species listed according to the Texel/Faial criteria will score poorly on at least some of Steps 1–4 of the process outlined above. We stress that this does not mean that these species do not need programmes of conservation and recovery. Some species may require them more urgently than listed species that do pass all four steps. The message is simply that the conservation and recovery plans ought to be developed and implemented outside the EcoQ – EcoQO framework.

3.4.2 Issues 3 and 4: Sea Mammals and Seabirds

Table 3.4.2.1. Marine mammals and birds. Metrics were graded against those features considered to be qualities of good EcoQOs (ICES, 2001a). Dark-shaded rectangles fully match the criterion; lightly-shaded rectangles do not fuly match the criterion and further improvements (where considered possible) are discussed in the section indicated.

Ecological quality element	a) Under-standable	b) Sensitive	c) Linked	d) Low error	e) Responsive	f) Measurable	g) Time series	h) Wider environment
Utilisation of seal breeding sites in the North Sea		3.4.2.1 a)	3.4.2.1 a)		3.4.2.1 a)			
Mercury concentrations in seabird eggs and feathers			3.4.2.2 a)		3.4.2.2 a)			
Organochlorine concentrations in seabird eggs								
Plastic particles in stomachs of seabirds			3.4.2.2 c)					
Local sandeel availability to black-legged kittiwakes[1]		3.4.2.2 d)						
Seabird populations trends as an index of seabird community health		3.4.2.2 e)	3.4.2.2 e)	3.4.2.2 e)	3.4.2.2 e)			

[1] The metric proposed was of black-legged kittiwake breeding success as an indicator of local sandeel availability to black-legged kittiwakes; this metric is evaluated in this table.

Notes:

a) Relatively easy to understand by non-scientists and those who will decide on their use.

b) Sensitive to a manageable human activity.

c) Relatively tightly linked in time to that activity.

d) Easily and accurately measured, with a low error rate.

e) Responsive primarily to a human activity, with low responsiveness to other causes of change.

f) Measurable over a large proportion of the area to which the EcoQ metric is to apply.

g) Based on an existing body or time series of data to allow a realistic setting of objectives

h) Relate to wider environmental condition.

3.4.2.1 Marine mammals

a) Utilisation of seal breeding sites in the North Sea

As discussed in ICES (2001a), this metric would be very easily understood by the non-scientist and in most areas supported by the wider public. The factors underlying seal breeding site distribution are not researched, but are certainly partially responsive to human disturbance (the largest rookeries are in undisturbed, remote areas). The linkages between distribution and other factors are less well known. Without such knowledge, certainty in management actions will be low. Research to explore the underlying factors could be encouraged.

As with breeding numbers, the Working Group on Marine Manmmal Population Dynamics and Habitats (WGMMPH) is the only existing international group in a position to compile distributional data. Therefore, we recommend that ICES be tasked to lead the scientific implementation of this EcoQO and integrate the work with the collation of breeding numbers around the North Sea. As specific tasks, ICES could publish a) a standardised seal censusing manual, and b) an annual report on the state of North Sea seal populations.

3.4.2.2 Seabirds

a) Mercury concentrations in seabird eggs and feathers

Mercury input to the marine ecosystem is predominantly anthropogenic and there is a very good historical time series based on skins in museums. The persistence of mercury in marine food webs means that there would be a lag between taking action to reduce anthropogenic input and the response in seabird eggs and feathers. Allowance would also need to be made for species and local variations in concentrations (ICES, 2001a). Research to understand this variation may improve the performance of this EcoQO.

ICES, with its track record on advice on seabird and on contaminant issues, including on mercury in other biota and sediments, would be well placed to coordinate a North Sea monitoring programme, publish monitoring standards, and produce reports on mercury concentrations in eggs and feathers. There would be little point in producing such reports annually, but a 2- or 5-year review cycle might be appropriate.

b) Organochlorine concentrations in seabird eggs

Current monitoring programmes in the Wadden Sea have tested and standardised procedures under the aegis of the Trilateral Monitoring and Assessment Programme (TMAP). This programme could be relatively easily expanded to cover other coasts of the North Sea. The remaining shortcoming of this EcoQO is due to the long persistence of many organochlorine compounds – it will take many years before they disappear from the marine environment even if all discharges stopped immediately

As with mercury, ICES, with its track record on advice on seabird and on contaminant issues including on organochlorines in other biota and sediments would be well placed to coordinate a North Sea monitoring programme, publish monitoring standards, and produce reports on organochlorine concentrations in seabird eggs. There would be little point in producing such reports annually, but a 2- or 5-year review cycle might be appropriate.

c) Plastic particles in stomachs of seabirds

Since the ICES recommendations were published in 2001 (ICES, 2001a), van Franeker and Meijboom (2002) have conducted a pilot project on the Netherlands coasts on plastics in the stomachs of northern fulmars. They concluded that it is feasible for the Netherlands to start an annual monitoring programme of marine litter using stomach contents of beach-washed northern fulmars. They commended such monitoring as it provides sound information on marine litter abundance in the southern North Sea and would be relatively inexpensive when conducted alongside the current Dutch beached bird survey. The annual sample size for such a programme is ± 40 northern fulmars from Dutch beaches.

In relation to Ecological Quality Objectives for the North Sea, van Franeker and Meijboom (2002) recommended supporting a pilot project of a North Sea-wide study of northern fulmar stomach contents. To this end, they are seeking a partnership in the planning of an EU Interreg IIIb proposal 'Save the North Sea', which focuses on marine litter and is coordinated by the Keep Sweden Tidy Foundation.

Thus, some of the shortcomings (for instance, knowledge on variance between samples) identified by ICES (2001a) have been addressed. Coordination in the North Sea would run alongside whatever programme is established to monitor beached birds. ICES could compile and publish results through the Working Group on Seabird Ecology (WGSE).

d) Local sandeel availability to black-legged kittiwakes (black-legged kittiwake breeding success)

Black-legged kittiwake breeding success is sensitive to changes in food supply within their feeding area, but the food supply (sandeel almost exclusively in some areas) is only partially responsive to fishing by humans. It is thus not tightly linked to a human activity or necessarily particularly responsive to fisheries management. These weaknesses in the EcoQO are unlikely to be improved much by further research.

Monitoring of black-legged kittiwake breeding performance is already undertaken at a good sample of UK colonies in the North Sea using standardised methods (Walsh *et al.*, 1995). Advice has already been provided by ICES on appropriate levels of black-legged kittiwake breeding performance (ICES, 2000b). The species also nests at Helgoland, on a Dutch gas platform and at sites in southern Norway. It would be very easy to add these localities to the existing scheme and report on them in the current annual UK/Ireland seabird monitoring report. Advice on black-legged kittiwake breeding success might be included within advice on sandeel stocks supplied to fisheries managers.

e) Seabird population trends as an index of seabird community health

This "EcoQ" is in fact a multiple one, as EcoQOs could be established for each seabird species monitored reliably in the North Sea. The relationship between breeding numbers and human activities is not well known. However, when ICES advised on this potential EcoQ in 2001, it considered that it would be suitable to act as a trigger for further research to determine whether manageable human activities are the cause of any actual decline. In the meantime, research on aspects of the interaction between humans and seabirds will continue.

Such monitoring of breeding numbers is conducted for selected sites on UK coasts using standard methods (Walsh *et al.*, 1995); for some species this monitoring is of the majority of the UK North Sea population. Monitoring is also conducted on other North Sea coasts. These monitoring programme results are published separately and could usefully be bought together, perhaps through WGSE.

3.4.3 Issue 5: Fish communities

a) Changes in the proportion of large fish, and hence the average weight and average maximum length of the fish community

This ecological quality element basically consists of two metrics:

- average weight of a fish in the community;
- average maximum length of a fish in the community.

Although both metrics are considered to be indicators of the proportion of large fish in the community, it should be realized that they represent different aspects of the community and are complementary in that respect. The average weight of the community represents changes in the size structure of the community, whereas the average maximum length represents changes in the species composition (Piet, 2001).

At the last meeting of WGECO (ICES, 2001b), both metrics were evaluated relative to a number of criteria that were deemed desirable in an EcoQ metric. In this evaluation, the same two criteria were considered to be not fully addressed by either of the metrics: (1) high response to signal from human activity compared with variation induced by other factors, and (2) tight linkage in time to that activity.

In addressing the concerns pertaining to the first criterion, two parts can be distinguished:

1) the degree to which the metric is representative of the changes occurring in the community (i.e., the proportion of large fish);

2) the relationship between human activity and that aspect of the community.

Several metrics have been proposed that are able to detect changes in the size structure. Most of these metrics failed on the other criteria and did not show a better signal-to-noise ratio (ICES, 2001b; Piet, 2001). Therefore, the average weight was found to perform best. The robustness and unambiguousness of this indicator is underlined by the fact that several surveys that are conducted throughout the North Sea over different periods of time show the same signal (see Section 5.3.2, below). Thus the average weight can be considered the best metric to show changes in the size structure of the fish community. Further work on this metric (see Section 5.3.2, below) showed that the signal-to-noise ratio might be further improved by selecting only those species (i.e., the demersal assemblage) that are adequately sampled by the gear. It should be realized that the selection of a subset of the community or the choice of survey (and therefore gear) have implications for the setting of the reference, current, and target levels as the metric only reflects the fish community as represented by the sampling technique and/or species selection. However, the consistency among surveys and other evidence gives confidence that the metric is a true reflection of the status of the fish community.

Another aspect of the fish community is the species composition. As large and long-lived species suffer a higher mortality the proportion of these species can be expected to decrease in an exploited community, thereby changing the species composition. By weighting a species-specific life history characteristic with the proportion of that species in the community, the change in species composition dependent on life-history characteristics can be quantified. For this, several life history characteristics exist that to a greater or lesser degree are related. Average maximum length expresses only the change in species composition. The reason for choosing this metric was that this life history characteristic was available for most species and that it appeared to be relatively sensitive.

Although this does not apply for each of these metrics separately, the combination of a metric that reflects changes in size structure (average weight) and one that reflects changes in species composition (average maximum length) does permit discrimination between a treatment that allows individuals of exploited species to grow larger and one that changes the species composition towards a higher proportion of large and long-lived species.

In exploited fish assemblages, larger fish generally suffer higher fishing mortality than smaller individuals and the size distribution becomes skewed towards the smaller end of the spectrum (Pope and Knights, 1982; Pope *et al.,* 1988; Murawski and Idoine, 1992). What is still unexplored is quantification of the association between fishing effort and aspects of the community that need to be preserved. This would allow an answer on questions such as what level of effort a specific community can tolerate without compromising its main characteristics or, in case changes have occurred, what measures should be taken to restore the community to a desired state.

A first attempt to further explore the relationship between fishing effort and community characteristics is done for different parts of the North Sea in Sections 5.3.1 and 5.3.2. The synthesis of these results (Section 5.4) shows that there is no straightforward relationship and an evaluation the results reveals two factors that hampered the analysis. In order to make progress on further operationalizing this element, the following factors need to be addressed:

1) Fishing effort data: long-term effort data with a high spatial resolution of all international fleets that fish in the area need to be available. At present, the following shortcomings apply to the data:

- they are only available for a few of the most recent years;
- they do not always include all fleets from all nationalities;
- they are expressed in a measure (i.e., days-at-sea or hours fished) that is not representative of true fishing effort and do not allow to distinguish between the impact of different gears (i.e., otter trawl versus beam trawl);
- data are at a relatively coarse resolution (ICES rectangles) which may not be a problem when assessing the effects on a mobile fish community, but will certainly apply when assessing the impact on the benthic community.

A way forward would be to use the data that are collected by the satellite-based monitoring programme that records the activities of all EU-based fishing for enforcement purposes. In this programme, all vessels larger than 24 m are monitored at an interval of about every two hours and at high (< 100 m) spatial resolution. These data are available at the national inspectorates and are confidential but should become available for scientific purposes.

2) Evaluation of management measures: opportunities to assess the effect of fishing on communities arise when measures are taken that (partly) close areas for fishing. In assessing the subsequent changes in the community and attributing this to the change in fishing activities, a number of difficulties arise:

- natural variation: in order to be able to account for natural variation, comparable areas are necessary in which no changes in effort occurred;
- because fish often cover relatively large distances most (semi-) closed areas will not be large enough to be able to detect change.

Provided that they are part of a properly designed experiment with areas that can be used as a reference, the closing of areas for fishing may be helpful in providing insight into the management responses needed to modify current levels. It should be realized that they will not result into the protection of the fish community unless these measures are applied together with effort reductions.

As was identified during last year's WGECO meeting, it is impossible to determine a reference level (i.e., where anthropogenic influence is minimal) since monitoring commenced long after pristine conditions were perturbed. Current levels, however, are adequately determined by several surveys and the length of the time series of many of these surveys already provides enough information to set a target level for these metrics. Considering the extent of the management measures that are probably necessary to reach these target levels, it is hardly realistic to aim for levels closer to the presumed reference level.

For all fish community metrics, there is a useful role for modelling, to make fuller use of historical data in identifying appropriate reference levels, and in helping to partition the role of various natural and anthropogenic factors in causing changes in the metrics.

3.4.4 Issue 6: Benthic communities

The current EcoQ element (with the background document) states:

(m) Changes/kills in zoobenthos in relation to eutrophication: There should be no kills in benthic animal species as a result of oxygen deficiency and/or toxic phytoplankton species.

Of the four ecological quality elements, two are proposed which do not form part of the pilot scheme. They are:

(o) Density of sensitive (e.g., fragile) species

(p) Density of opportunistic species

WGECO considered these together, because apart from the criteria used for selection of the species to be considered under each, the approaches are identical. At the last meeting of WGECO (ICES, 2001b), we developed a framework for rigorously testing EcoQOs and applied it to a large number of possible metrics of benthic ecosystem status. This objective testing regime led us to conclude that, with existing knowledge, the only possible EcoQO that could be made operational for the benthos was one based on the abundance of sensitive/indicator taxa. We are therefore heartened to see that the Bergen Declaration follows this logic by adopting the density of sensitive and opportunistic taxa as EcoQOs for benthic communities.

However, it is now urgent that consideration be given to the development of robust and objective criteria for selecting species for the two lists and establishing the criteria that should be used in selecting the baseline (reference) levels for these species in the system.

To develop robust and objective criteria for selecting "sensitive" (or "fragile") and "opportunistic" species, it is necessary both to examine observational data and to derive independent criteria. The former requires the assembly of the available data on the distribution and abundance of benthic taxa in the North Sea and for this to be related to the distribution of impacting activities. This would require a formal meta-analysis of the various data sets rather than the all too frequent subjective commentary. The second approach requires objective definitions of "sensitive" (or "fragile") and "opportunistic" to be produced and species to be reviewed against these criteria. Last year's report (ICES, 2001b) provides thoughts on how to proceed in developing such objective definitions. To avoid circularity, species should be tested against these criteria using carefully planned, *a priori* comparisons (ICES, 2001b). Moreover, it is important to bear in mind that EcoQOs should be related to specific human activities. If management responses are to be targeted on specific activities, then the criteria must relate to specific, not generic, threats.

It is also likely that a "sensitive" or "opportunistic" species will be related to a specific activity or context. A species (e.g., bivalve) could be sensitive to an activity such as dredging, but less sensitive to other activities (e.g., eutrophication, SCUBA diving). Using sensitive species to identify the impact of anthropogenic activities could be a problem if this activity already decreased or even had extirpated the sensitive species from an area. The same principle can be applied for the opportunistic species, but with the opposite trends.

Within the EcoQ – EcoQO framework, a reference level must be specified. Notwithstanding past WGECO observations on the inappropriateness of the "pristine state" as the default reference point (ICES, 2001b), this approach continues to have support by some proponents of the EcoQ – EcoQO framework (OSPAR, 2002). If the reference level is to be the undisturbed state, it will be necessary to characterise a benthic community before the damaging activity. Most of the time this is impossible, and alternative approaches are required (Glasby, 1997). These may require developing an experimental design (potentially impacted site vs. natural sites), or using the available data in a meta-analysis. Both of these tasks require directed scientific efforts by skilled benthic ecologists, and are likely to require some original research, not just repackaging existing information.

Once a reference point is identified, it will not be straightforward to monitor status relative to it. Issues of the statistical power of monitoring data will be very important, and weak tests may be hard to avoid (ICES, 2001c). Spatial and temporal variations of sensitive or opportunistic species must be estimated properly, too, if population trends and responses to management actions are to be detected. Natural variation is high in benthic communities and could lead to a false interpretation of the change in the benthic sensitive or opportunistic species. To distinguish an increase in opportunistic species (or a decrease in sensitive species), comparisons should be done with at least two control areas, because of the natural variability. If the pattern is compared with only one control area, the result will be confounded between the activity and a site effect. Again, confusion will be present. A monitoring programme of benthic communities (see project REBENT–Réseau benthique, in Brittany, France) in many sites (many spatial scales) and few times per year (temporal scale) for some species is one approach to addressing these concerns.

The temporal and spatial scales will also be important for EcoQOs using sensitive or opportunistic benthic species. What is the relevant scale to sample? How important is the spatial extent of the impact? The relevant scales of sampling to detect a particular impact cannot simply be implied by modelling or monitoring physical and chemical variables, as has been done in and recommended by many previous studies (e.g., Spellerberg, 1991). Sampling at several scales is important given that sampling at the wrong scale may result in failure to detect an impact and given that populations may respond to disturbances in different ways at different spatial scales (Bishop *et al.,* in press).

3.4.5 Issue 8: Habitats

A single ecological quality element is proposed for the habitats issue: (s) Restore and/or maintain habitat quality.

This is a very laudable expression of intent but in this form is far from being operational. Amongst the key constraints at this time are:

- the lack of an agreed framework of habitat classifications for European marine habitats (see ICES, 2000a, Section 3 for a detailed discussion of this topic).

There is therefore still an urgent need to advance the marine habitats portion of EUNIS. The work needed to advance the marine habitats portion of EUNIS has been outlined in WGMHM 2001, and WGECO concurs with that approach:

- The treatment of "habitat quality" as a singular term implies it is the quality of the habitat that is being viewed in some integrated sense. If "habitat quality" is being thought of in such a conceptually unitary way, the concept cannot be made operational. It is only possible to measure "habitat" on a site-by-site basis, and there is no scientific way to calibrate "quality" across different sites with different habitat features. If the intent was to keep the habitat at every individual site from being degraded, and restoring all sites that were degraded to a healthy condition, it is still impossible to operationalize. The status of all habitats is unknown and unknowable, and it is impossible to know how every habitat is changing. It is also impossible to interpret every change in habitat as either a "recovery" or a further degradation. This EcoQ element has to be restated into a form where realistic measurement programmes of habitat features and sites would be adequate to track status, change, and compliance with reference levels, once set.

- This definition would seem to exclude any deterioration of any habitat anywhere, at any time, no matter what the societal benefit of such an action – this is naïve and flies in the face of the provisions of the BCD which allow development provided the societal and economic benefits outweigh the negative environmental effects.

When this is converted into an EcoQO, consideration needs to be given to incorporating some quantitative conservation limits (like B_{lim} for commercial fish stocks), rather than the ABSOLUTE standard of "no change" from whatever the status of a habitat was at the commencement of monitoring. These would protect habitats effectively from loss or serious damage, but still allow development and sustainable utilisation of environmental goods and services.

- What habitats are to be restored and to what state are they to be restored? The first consideration is again naïve and impossible. It presupposes knowledge of previous (pristine? or just "healthier") states of all habitats, as well as simply mandating an imperative that all altered habitats be returned to some condition that will be specified somehow. It also presupposes the capability to engineer habitats at will, and the existence of some rules for deciding which habitats need restoration and which ones do not. The knowledge and tools do not exist and, most importantly, it has not been demonstrated to the satisfaction of, at least, WGECO that a "healthy" North Sea requires that all disturbed habitats be restored. The second consideration ignores the realities of open dynamic ecosystems like the marine environment, where active restoration programmes rarely succeed or make economic sense (Frid and Clark, 1999; Hawkins *et al.,* 1999), as well as presupposing either that pristine states of all habitats are known or else some rules exist for deciding how close to pristine—or how far from current levels or perturbation—it is necessary for restoration to take a site.

For all those shortcomings, there are science undertakings that could be done to provide a sounder knowledge base for moving towards making the EcoQ element operational. We do not recommend those undertakings for the sake of making this EcoQ element operational, however (although we may recommend some of them, on various scales, for other reasons). Rather, we recommend a reworking of the EcoQ element itself, into something both more feasible in the field and more conceptually tractable.

There are several considerations that might help guide the reworking of the EcoQ element. Restoration ecology is a relatively new discipline and experience in marine systems lags behind that in the terrestrial environment. However, it would also appear that marine systems, at least open coastal systems, have a great capacity for self repair once the *impacting activity is removed. Some effort needs to be afforded in establishing criteria for assigning value (on several dimensions, including ecological role, human needs, ethics and aesthetics, etc.) to habitats, and, using those criteria, establishing a priority list of habitats. For the priority habitats, it will be necessary to gather information on what a "healthy" condition is for the habitat type and the best way to achieve the "healthy system" – active intervention (restoration) or passive monitoring only.*

- How does one measure a multivariate quality such as 'habitat quality'? Methods commonly used to quantify habitat status (Gauch, 1982; Jongman *et al.,* 1987) can produced axes where "quality" can be identified for one habitat type. However the same axes are not applicable to all habitats. Either different metrics of quality will be required for each habitat, or methods will have to be discovered to calibrate the position of the "healthy" state consistently across numbers of habitat axes. Even if one can find a measure for this (a multivariate statistical parameter, for example), there is currently no justification to assume that the measure will be sensitive to and vary in a predictable way in response to specific human impacts, which might in turn be managed.

Considerable research effort will be required to establish appropriate metrics of habitat quality. These are likely to be applicable to a limited number of habitats each; therefore, a considerable number will be required to cover the habitat types found in the North Sea.

References

Bishop, M.J., Underwood, A.J., and Archambault, P. Sewage and environmental impacts on rocky shores: necessity of identifying relative spatial scales. Mar Ecol. Prog. Ser. (in press M 4108).

Burgman, M.A., Ferson, S., and Akcakaya, H.R. 1992. Population Viability Analysis: methods for estimating the risk of extinction of biological populations. Chapman Hall, New York.

Burgman, M.A., Ferson, S., and Akcakaya, H.R. 1993. Risk Assessment in Conservation Biology. Chapman and Hall. New York.

FAO. 1999. A Review of the Applicability of the CITES Criteria of Listing Endangered Species for marine fish. FAO Technical Report, 356: 76 pp.

Frid, C.L.J., and Clark, S. 1999. Restoring aquatic ecosystems. Aquatic Conservation: Marine and Freshwater Ecosystems, 9: 1–4.

Gauch, H.G. 1982. Multivariate Analysis in Community Ecology. Cambridge University Press. Cambridge, UK. 298 pp.

Glasby, T.M. 1997. Analysing data from post-impact studies using asymetrical analyses of variance: a case study of epibiota on marinas. Australian Journal of Ecology, 22: 448–459.

Glasby, T.M., and Underwood, A.J. 1996. Sampling to differentiate between pulse and press perturbations Environmental Monitoring and Assessment, 42: 241–252.

Gubbay, S. 2001. Development of Ecological Quality Objectives for the North Sea – Threatened and declining species. Working Paper for WGECO 2001.

Hawkins, S.J., Allen, J.R. and Bray, S. 1999. Restoration of temperate marine and coastal ecosystems: nudging nature. Aquatic Conservation: Marine and Freshwater Ecosystems, 9: 23–46.

Hutchings, J.A. 2001. Conservation biology of marine fishes: perceptions and caveats regarding assessment of extinction risk. Canadian Journal of Fisheries and Aquatic Science, 58: 108–121.

ICES. 2000a. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 2000/ACME:02. 93 pp.

ICES. 2000b. Report of the ICES Advisory Committee on the Marine Environment, 1999. ICES Cooperative Research Report, 239: 7–10.

ICES. 2001a. Report of the ICES Advisory Committee on Ecosystems, 2001. ICES Cooperative Research Report, 249: 15–59.

ICES. 2001b. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 2001/ACME:09.

ICES. 2001c. Report of the ICES Advisory Committee on the Marine Environment, 2001. ICES Cooperative Research Report, 248:173–179.

IUCN. 1999. IUCN/SSC Criterion Review Working Group. IUCN Red List Criteria Review - provisional report: Draft of the proposed changes and recommendations. Species 31–32: 43–57.

IUCN. 2001. IUCN Categories and Criteria vs 3.1. 35 pp.

Jongman, R.H.G., ter Braak, C.J.F., and van Tongeren, O.F.R. 1987. Data analysis in community and landscape ecology. Pudoc-Wageningen, Den Haag. 299 pp.

Mace, G.M., and Stuart, S.N. 1994. Draft IUCN Categories, Version 2.2. Species 21–22; 13–24.

Murawski, S.A., and Idoine, J.S. 1989. Multispecies size composition: a conservative property of exploited fishery systems. Northwest Atlantic Fisheries Organisation Document, 89/76, 12 pp.

Murawski, S.A., and Idoine, J.S. 1992. Multispecies size composition: a conservative property of exploited fishery systems. Journal of Northwest Atlantic Fisheries Science, 14: 79–85.

Musick, J. 1998. Endangered marine fishes: Criteria and identification of North American stocks at risk. Fisheries, 23(2): 28–30.

Musick, J. 1999. Criteria to define extinction risk in marine fishes. Fisheries, 24(12): 6–14.

OSPAR. 2000. Criteria for the selection of species and habitats which need to be protected. ASMO 00/9/3-E. 19pp.

OSPAR. 2002. Background document on the development (within the OSPAR framework) of Ecological Quality Objectives (EcoQOs) for the North Sea. Paper to the meeting of the Committee of North Sea Senior Officials, Oslo, 9–10 January 2002. DOC CONSSO JAN 02/3/24.

Piet, G.J. 2001. Development of ecological quality objectives for the North Sea fish community. Working paper at the 2001 WGECO meeting.

Pope, J.G., and Knights, B.J. 1982. Comparison of the length distributions of combined catches of all demersal fishes in surveys in the North Sea and at Faroe Bank. *In* Multispecies approaches to fisheries management. Ed. by M.C. Mercer. Canadian Special Publication on Fisheries and Aquatic Science, 59: 116–118.

Pope, J.G., Stokes, T.K., Murawski, S.A., and Idoine, J.S. 1988. A comparison of fish size-composition in the North Sea and on Georges Bank. *In* Ecodynamics - contributions to theoretical ecology, pp. 146–152. Ed. by W. Wolff, C.-J. Soeder, and F.R. Drepper. Springer-Verlag, Berlin.

Powles, H., Bradford, M.J., Bradford, R.G., Doubleday, W.G., Innes, S., and Levings C.D. 2000. Assessing and protecting endangered marine species. ICES Journal of Marine Science, 57: 669–676.

Spellerberg, I.F. 1991. Monitoring ecological change. Cambridge University Press, Cambridge. 334 pp.

Van Franeker, J.A., and Meijboom, A. 2002. Litter NSV - marine litter monitoring by northern fulmars (a pilot study). Alterra-rapport 401, Alterra, Wageningen. 72 pp.

Walsh, P.M., Halley, D.J., Harris, M.P., del Nevo, A., Sim, I.M.W., and Tasker, M.L. 1995. Seabird monitoring handbook for Britain and Ireland. Joint Nature Conservation Committee, Royal Society for the Protection of Birds, Institute of Terrestrial Ecology and Seabird Group, Peterborough, UK.

3.5 Summary and Conclusions

The commitment to proceed with an EcoQ – EcoQO framework for conservation and protection of the North Sea ecosystem is a significant step in the implementation of an Ecosystem Approach. However, a great deal of work is needed to deliver the promise of the commitments: work on short-term, medium-term, and long-term scales.

The short-term work is required to proceed with the EcoQOs specified in Table B, Annex 3 of the Bergen Declaration. Although not all the EcoQOs are ideal, according to the objective criteria for operational EcoQOs developed last year, they provide a reasonable suite of EcoQOs for the pilot project described in paragraph 4iv) of the Declaration. Our evaluation indicates that some additional work is needed to proceed with a few of them, but there is adequate science available at present to proceed with most of them. It is important, though, that the monitoring and evaluation in the pilot project be done carefully, and use standards to which we provide guidance in Section 3.3.

The medium-term work is required to be able to set EcoQOs for the EcoQ elements in Annex 3, Table A, that currently lack EcoQOs in Table B. In some cases we conclude that the scientific information is adequate for setting EcoQOs at present, and-only societal choices about reference levels prevent moving ahead (e.g., Section 3.4.2). For some other EcoQ elements, however, the scientific basis is very far from adequate for setting EcoQOs (ex Section 3.4.5). We provide specific guidance about the science tasks necessary to fill in the scientific foundation for EcoQOs. In some cases, though, for instance the habitat EcoQ element, there are so many serious gaps in the science that we conclude that the commitment in paragraph 4iii) that “By 2004, EcoQOs for the remaining elements will, in the same way, be developed and applied within the framework of OSPAR …” is completely unrealistic. We stress that setting EcoQOs prematurely, on an inadequate scientific foundation, is as likely to be step backwards as a step ahead. Premature action might lead to management actions and monitoring and evaluation programmes that are doomed to be inconclusive, ineffective at improving the environmental health of the North Sea, and costly to managers, resource users, and the scientific community.

The long-term work is necessary to truly deliver “an Ecosystem Approach for the North Sea” with “a coherent and integrated set of Ecological Quality Objectives”. Even if the pilot project is a success, no one should be complacent about having secured the environmental health of the North Sea. For many important properties of the North Sea (or any marine) ecosystem, the science is far from ready to provide a basis for setting EcoQs and EcoQOs. If the EcoQ – EcoQO framework is effective, it provides a welcome way to focus the science and advisory tasks ahead of us. It only provides a focus and framework, however, and not a shortcut. Sound science will remain a prerequisite for sound management and decision-making.

Recommendations

With regard to the EcoQ issue of threatened and declining species (Issue 2), ICES should work with OSPAR and other experts to:

- identify from the list of species designated as threatened and declining, species that would be particularly appropriate for developing robust and effective EcoQOs (see Bergen Declaration, Annex 3, Table A);
- provide the scientific basis for setting reference levels for the EcoQOs;
- provide the scientific and statistical basis for estimating current levels and for the monitoring that would be part of the pilot project.

With regard to the EcoQ issue on fish communities (Issue 5):

- WGECO should consolidate the scientific basis for the EcoQ elements and reference levels currently proposed for fish communities;
- ICES and/or WGECO should further quantify the relationship between fishing effort and the two metrics of fish communities included in Table A, Annex 3;
- WGECO should continue to develop and evaluate candidate metrics of fish communities, particularly with regard to factors such as spatial integrity and ecological functionality that we concluded last year were not currently addressable with robust and effective EcoQOs.

With regard to the EcoQ issue on benthic communities (Issue 6):

- ICES should engage with OSPAR in the development of objective, empirical criteria for the selection of sensitive and opportunistic benthic species;
- ICES should contribute its expertise to the setting of appropriate levels of sensitive and opportunistic benthic species for use in setting operational EcoQOs.

With regard to the EcoQ issue on habitats (Issue 8):

- ICES should engage with the European Environment Agency and OSPAR in furthering the development of a marine habitat classification scheme for habitats in the OSPAR region;
- ICES should contribute expertise to the detailed mapping of marine habitats in the OSPAR region;
- WGECO, at a meeting in the near future, should consider the question of what properties of "habitats" might eventually be usable as metrics by which to measure efforts to "restore and/or maintain habitat quality". If the results are not promising, provide the basis for discussions among ICES, OSPAR, and the larger scientific and management communities on alternative EcoQ elements to address the issue of "Habitats".

Justification

The principal barriers to advancing EcoQOs for sensitive and opportunistic benthic species and for the maintenance of habitats are criteria for their selection. The development of a logical framework for doing this clearly requires the input of scientific understanding as well as societal values.

4 QUANTIFY THE RELATIVE ROLE OF FISHING AND OTHER HUMAN ACTIVITIES ON THE DYNAMICS OF THE MARINE ECOSYSTEM

"quantify the relative role of fishing activity on dynamics of the marine ecosystem and nutrient turnover, in comparison with other comparable human activities such as marine disposal, and mineral extraction, to the extent possible."

4.1 Introduction

Annex V of the OSPAR Convention and the Convention on Biological Diversity require the protection of the maritime area against the adverse effects of human activities so as to safeguard human health and to conserve marine ecosystems. As a result, Contracting Parties are expected to adopt programmes and measures for the control of relevant human activities. The recommended criteria used to select human activities are sufficiently wide-ranging to encompass a large number of human activities in estuarine, coastal, and offshore waters in the OSPAR region. Thus, in order to make progress with management actions for the most important issues, some prioritisation of these human activities is required. The results of such a ranking exercise could also have implications for national marine monitoring and management, where the increased focus on local and regional assessment requires knowledge of the most important issues for priority action.

WGECO has been asked to evaluate the impacts of human activities on ecosystem dynamics and nutrient turnover. This topic has already been discussed by WGECO (ICES, 1992a) when preliminary assessments were made of the spatial extent of a range of human activities. There was, however, no comparison of their impact on the ecosystem in a comparable way, and so when re-attempting this work now we have interpreted changes in ecosystem dynamics as changes to mortality and production, although it is recognised that there are many other aspects of the ecosystem, such as the provision of goods and services, which are not considered. While we recognise the weaknesses in this interpretation, adequate data on other aspects of dynamics were not available. Our approach has been to reach some initial conclusions and to describe a process for completing these comparisons, but completion of the process requires the acquisition of additional data.

It is also important to bear in mind that human activities rarely act independently. To some extent, it is artificial to prioritise separate human activities and more important to consider the cumulative or in-combination effects of multiple activities, and how they act together.

4.1.1 The approach of the working group to the ToR

Table 4.1.1.1 summarises the conclusions of a detailed evaluation and analysis conducted by the National Institute for Coastal and Marine Management, Delft, the Netherlands (Resource Analysis, 1998). Before deciding how to proceed with a detailed quantification of the relative role of human activities on the dynamics of the marine ecosystem, it is necessary to review the methods and approach used by Resource Analysis (1998) for the North Sea. Subsequent sections of the report will highlight a suitable approach to further develop this prioritisation process by quantifying the impacts of certain activities, especially in relation to the role of fishing, dredging, and disposal on seabed disturbance and nutrient turnover. We have limited the analysis of data to the North Sea.

4.1.2 Outcome of OSPAR prioritisation

The recent OSPAR North Sea Quality Status Report (OSPAR, 2000) provided an overall assessment of 32 human activities or "pressures" prioritised into four classes. A structured prioritisation method was used and each pressure was evaluated against a hierarchical set of criteria. Although the report states that the division into classes was robust, it was not clear to the working group how this division was determined. Class A was considered of highest impact, class B upper intermediate impact, class C lower intermediate impact, and class D lowest impact. The summary conclusions of this analysis are reproduced in Table 4.1.1.1.

4.1.3 The Multi Criteria Approach (Resource Analysis, 1998)

The overall assessment by OSPAR of the Greater North Sea was based on the effects of human activities or pressures on the full range of socio-economic and ecological issues related to the area. Given the complex interactions and processes involved in this evaluation, such effects cannot be expressed in a single scale. Multi Criteria Analysis (MCA) is a set of techniques or procedures that provides a ranking of different indicators which are measured on scales that have different units. MCA also requires that an explicit and objective basis for structuring the evaluation is prepared. This MCA approach helps to achieve consensus on the final prioritised list by coordinating and presenting data, enabling multi-user input of views and encouraging discussion.

The MCA analysis was based on a selection of environmental criteria within a hierarchical structure, where the main objective was sustainable use. Nine criteria were identified within this structure: ecology, chemical environment, physical habitat, recreation, fisheries and mariculture in relation to the coastal zone, and ecology, chemical environment, physical habitat, and fisheries in relation to the open sea.

The effects of different human activities on these environmental criteria were measured using three different aspects: severity, spatial scale, and recovery time. A nine-point scale was used to score each of these aspects of activity, and to influence their relative importance in the final evaluation, the outcomes were each weighted by the following factors: severity 0.6; spatial scale 0.3; recovery time 0.1.

Two workshops were used to complete the evaluations. Representatives from eight countries participated and completed a matrix of 27 indicators (3 aspects of 9 criteria) for 33 human activities, i.e., 27 × 33=891 scores. Each of these scores was derived from a question such as: How do you rate the (severity / spatial scale / recovery time) of (one of the 32 human impacts) on (one of the nine ecological criteria). Final scores for each activity were based on group averaged ranking. Activities were divided into four classes, but there was no statistical basis for this division (Resource Analysis, 1998).

4.1.4 Appraisal of the technique

The advantage of the MCA approach is that it comprises a comprehensive process, which is relatively clear and intuitive. There is also an iterative process built into the final assessment so that there is an opportunity to revise initial scores to standardise between countries.

The disadvantage of the approach is that the final prioritisation is based on a numerical score for each human activity (Table 4.1.1.1), and yet the score is based on subjective assessments of the perceived impact of activities on biological resources. Persons completing the analysis must reach a conclusion, and derive a score of between 1 and 9 for the magnitude of the effect. Furthermore, this score must relate to all the coastal zone and open sea environments in the Greater North Sea. So for example, the severity of effect of "removal of target species by fisheries" must be judged for the entire North Sea coastal zone.

4.2 An alternative approach to the quantification of impacts

Since the MCA approach is based on the subjective assessments of the persons involved, it cannot be expected to yield consistent results. In order to develop a more objective approach based on a quantitative assessment, we here describe the spatial extent and level of impact of two widespread activities, bottom trawling and marine aggregate extraction, using a more objective method. This involves a comparison of the spatial extent of the impact, and a suggested approach for quantifying effects on population dynamics (in terms of total mortality and production). The approach focuses on comparison of short-term impacts (years), rather than addressing the longer-term consequences of habitat modification and removal by beam trawling or dredging.

Table 4.1.1.1. Priority classes of human pressures (reproduced from Resource Analysis, 1998).

Class*	Pressure	Score
A	Fisheries removal of target species	0.439
	Inputs from land: organic micro-pollutants	0.403
	Fisheries seabed disturbances	0.384
	Inputs from land: nutrients	0.341
	Fisheries effects of discards and mortality of non-target species	0.332
	Shipping: inputs of TBT & other anti-fouling substances	0.331
B	Offshore oil and gas industry: input of oil & PAHs	0.331
	Shipping: inputs of oil & PAHs	0.323
	Offshore oil and gas industry: input of other hazardous substances	0.295
	Inputs from land: heavy metals	0.279
	Inputs from land: oil & PAHs	0.267
	Shipping: introduction of alien species	0.234
	Shipping: input of other hazardous substances	0.233
	Mariculture: introduction of cultured specimens, alien species and diseases	0.228
	Inputs from land: microbiological pollution and organic material	0.224
C	Fisheries: input of litter (ghost nets)	0.223
	Offshore oil and gas industry: physical disturbance	0.217
	Shipping: input of litter	0.207
	Dredged material: dispersion of substances	0.176
	Military activities: (chemical) ammunition	0.175
	Engineering operations: constructions in the coastal zone	0.173
	Mariculture: input of chemicals	0.171
	Engineering operations: mineral extraction (sand, gravel)	0.167
	Mariculture: input of nutrients and organic material	0.162
	Dredged material: physical disturbance	0.156
	Inputs from land: radionuclides	0.152
D	Shipping: physical disturbance	0.150
	Recreation: input of litter	0.129
	Military activities: physical disturbance	0.129
	Recreation: physical disturbance	0.121
	Engineering operations: power cables and electromagnetic disturbances	0.115
	Dumping of inert material (wrecks, bottles)	0.110

* Human pressures are ranked according to their relative impact on the Greater North Sea ecosystem, including sustainable use. While the division in the four classes A–D was established firmly, ranking within classes was not considered to be significant. Class A = highest impact; Class B = upper intermediate impact; Class C = lower intermediate impact; Class D = lowest impact.

4.2.1 Marine aggregate dredging in the North Sea

Marine aggregate is generally dredged by trailer suction hopper dredgers, which produce shallow linear furrows approximately 1–3 m wide and 0.2–0.3 m deep. Repeated dredging by trailer dredgers can result in substantial lowering of the seabed across a wide area and this will be related to the frequency of dredging and the level of dredging intensity (Norden Andersen *et al.*, 1991). The most significant consequence of marine aggregate extraction is the removal of the substrate and the associated benthic fauna (ICES, 1992b; Kenny and Rees, 1994, 1996; Kenny *et al.*, 1998; Newell *et al.*, 1998; Desprez, 2000). Dredging can also lead to the production of plumes of suspended material from the draghead (Moran, 1991), from spillways on the vessel hopper, or from screening activities.

Electronic Monitoring Systems (EMS) in the UK automatically record the date, time, and position of all dredging activity every 30 seconds. Effort data (hours per year) are provided within areas of 100 m × 100 m. Dredging intensity data are published for all licences in terms of the area (km^2) in which < 5 and > 5 hours' dredging intensity per year took place.

The most significant consequence of marine aggregate extraction is the removal of the substrate and the associated benthic fauna (ICES, 1992b). Most studies on the effects of aggregate extraction have concentrated on establishing the post-dredging rates of macrobenthic recolonisation (Desprez and Duhamel, 1993; Kenny and Rees, 1994, 1996; Boyd and Rees, 2001; Desprez, 2000). These studies show that dredging causes an initial reduction in the abundance, species diversity, and biomass of the benthic community.

Using EMS data, Boyd and Rees (2001) conducted a survey to examine the impacts on the benthos arising from commercial aggregate extraction at sites east of the Isle of Wight (UK) which were subjected to different levels of dredging intensity. Samples from intensively dredged sediments showed reductions in numbers of species, biomass, species richness, and diversity. This study showed that the intensity of disturbance can influence the proportion of the total number of species that are affected, thereby prolonging the time-scale for re-establishment. Species such as *Sabellaria spinulosa* and some ascidians have been found to be more susceptible to disturbance from commercial aggregate extraction than others (Lees *et al.*, 1992; Boyd and Rees, 2001).

In a controlled study of the impacts of marine gravel extraction on the macrobenthos at a site off the east coast of England, Kenny and Rees (1994) showed that significant reductions had occurred in numbers of species (62 %), abundance (94 %), and the biomass (90 %) following the removal of 52,000 t of material by a trailer suction dredger. Similarly, a significant reduction in the abundance (72 %), biomass (80 %) and numbers of species (30 %) was reported two months after aggregate extraction on the Klaverbank (van Moorsel, 1993, 1994).

Statistics provided by ICES (2001) and the UK Crown Estate (2001) allow us to estimate the total area from which sand and gravel was removed in the North Sea south of 55 ºN, during 2000 (Table 4.2.1.1).

Table 4.2.1.1. Showing the estimated total area (km^2) dredged in 2000 by Belgium, Denmark, the Netherlands, Germany and UK, in the North Sea south of 55 ºN. UK total volume dredged is estimated by applying a conversion factor of 1.61 to the total tonnage removed (ICES, 2001). Total area dredged (UK) in 2000 is available from Crown Estate (2001). The area dredged by other European countries is estimated from the volume extracted (ICES, 2001), and applying an assumed mean dredge depth of 0.204 m, based on UK estimates.

	Tonnes	**m^3**		**Area dredged (km^2) (Crown Estate, 2001)**	**Estimated average depth dredged (m)**
UK	15,206,905	24,521,134	*	119.95	0.204
	* Conversion of UK tonnes/year to volume based on raising factor of 1.61				
		m^3		**area dredged (km^2)**	
Belgium		1,901,000			
Denmark		4,500,000			
Netherlands		25,400,000			
Germany		1,673,723			
TOTAL		33,474,723		163.85 *	
	* (assuming average depth dredged of 0.204 m)				
TOTAL AREA DREDGED				**283.8 km^2**	

There are currently no readily available estimates of the area dredged within continental European licence areas, although Dutch operators are expected to dredge in such a way that the seabed is gradually lowered within the total licence area (van Dalfsen, pers. comm.). Based on a figure of 0.204 m average depth of extraction, the area dredged by continental European countries was estimated at 163.85 km^2. In estimating a total area dredged for five countries of 283.8 km^2 in the North Sea during 2000, a number of assumptions were made. First, it was necessary to estimate the average depth of dredging in continental European licences from UK data, in order to convert the volume estimates of removal into a surface area impacted. This estimate also assumes that cargo capacity volumes provided by contributors to the Working

Group on the Effects of Extraction of Marine Sediments on the Marine Ecosystem (WGEXT) are equivalent to the volume occupied on the seabed by the same sand and/or gravel before extraction.

However, the impacts of dredging within these 283.8 km^2 are unlikely to be evenly distributed. The spatial and depth distribution of the resource will result in licences being exploited to varying depths. For example, although on the UK North Sea coast the total area dredged was 119.95 km^2, the most intensively dredged area (>5 hr per year per 100 m × 100 m) covered only 7.13 km^2 (Crown Estate, 2001). Clearly, this level of more intense dredging activity will result in more significant impacts to the seabed. Assuming that the dredger operated homogeneously within a 100 m × 100 m box for 5 hours, how much of a spatial impact does this represent? Assuming a dredge track width of 2.5 m, it would require 40 passes of a dredger (a distance of 4 km) to cover the entire 100 m × 100 m area. This distance could be covered in 1 hour at a speed of 2.5 knots, which suggests that over a 5-hour period at this speed the entire seafloor in a box 100 m × 100 m could be dredged approximately 5 times.

4.2.2 Beam trawling in the North Sea

There is now an extensive literature on the effects of trawling on the sea floor (Rijnsdorp *et al.*, 1998; Collie *et al.,* 2000). The impacts of beam trawling are determined by the penetration depth of the gear and the spatial distribution of beam trawl effort. WGECO commented on a substantial research programme, IMPACT II (Lindeboom, and de Groot, 1998), at its 1999 meeting (ICES, 2000), and has described and evaluated many of these impacts.

The beam trawl is a heavy gear that uses a series of chains to disturb the sediment surface in order to increase the catch rate of target species. Fishing mortality estimates of invertebrate populations due to 4-m beam trawls, 12-m beam trawls and otter trawls in the Dutch sector of the North Sea were provided by Lindeboom and de Groot (1998) and Bergman and van Santbrink (2000a, 2000b). The calculations were based on estimates of total mortality due to a single trawl pass, densities of benthic invertebrates by ICES quadrant, fishing effort by ICES rectangle, and the distribution of beam trawl effort based on effort micro-distribution data from a subset (n=25) of the Dutch fleet (Bergman and van Santbrink, 1997; Rijnsdorp *et al.,* 1998). The calculations did not consider intra-seasonal recruitment or sources of mortality other than trawling (i.e., natural mortality) and were reported in ICES (2000).

The annual fishing mortality in the larger-sized invertebrate populations varied from 7 % to 48 % due to trawl fisheries in the Dutch sector in 1994, with half the number of species showing values of >25 %. The 12-m beam trawl fisheries caused higher fishing mortalities than 4-m beam trawl and otter trawl fisheries. Only in species restricted to the coastal zone, where the 4-m beam trawl fishery is much more intensive than in offshore areas, were fishing mortalities relatively higher and might even exceed those due to 12-m beam trawl fisheries (Lindeboom and de Groot, 1998, p. 371). These estimates of annual mortality should be considered rough approximations as they depend heavily on the assumption of uniform spatial distribution of benthic invertebrates. Alternative assumptions would involve different options of species distributions, for example, invertebrate species having a low and uniform distribution across an ICES rectangle except for higher abundance (60 % of the total abundance in the area) in the 1/9th which is most heavily trawled (56.9 % of the effort). In addition, estimates of annual mortality depend on the highly uncertain total mortality estimates reported in Lindeboom and de Groot (1998). The annual fishing mortality estimates of benthic invertebrates ranged from 7 % to 33 % and are highly affected by the assumption of the type of distributions of the species and the fisheries within the rectangles (Lindeboom and de Groot, 1998).

In order to make a comparison with other human activities such as marine dredging, it is necessary to know the spatial distribution of the impacts attributable to the beam trawl fleet. Rijnsdorp *et al.* (1998) described the microdistribution of Dutch beam trawl effort in the southern North Sea, based on an automatic recording system. They estimated that during a four-year study period, in eight of the most heavily trawled rectangles in the southern North Sea, 5 % of the surface area was trawled less than once in 5 years, and 29 % less than once a year. Using revised estimates of the area of each rectangle, the surface area trawled more than five times in a year can be estimated at 10 %, and the surface area trawled more than ten times in a year can be estimated at 1.5 % (Table 4.2.2.1). These percentages correspond to 4160 km^2 for trawling intensities of >5 times, and 620 km^2 for intensities of >10 times (Table 4.2.2.1).

Table 4.2.2.1 Proportion of the surface area of the eight most heavily fished ICES rectangles trawled at a certain frequency (number of times one m^2 is trawled annually) (Rijnsdorp *et al.*, 1998). Geographic areas have been recalculated allowing for the variation in the areas of rectangles with latitude.

ICES RECTANGLE	rectangle total area (km^2)	proportion > 5 ×	km^2	proportion >10 ×	km^2
32F2	3827	4.9	187	0.8	31
34F3	3742	4.7	180	0	0
35F3	3699	9.6	367	0.3	11
36F4	3656	10.9	417	1.2	46
37F4	3612	5.7	218	0.3	11
37F5	3612	5.2	199	0	0
37F6	3612	19.3	739	2.4	92
38F6	3568	9.3	356	3.5	134
33F3	3785	16	612	0.7	27
35F4	3517	7.7	295	0.8	31
37F7	3612	15.4	590	6.2	237
Total	**40242**	**10**	**4160**	**1.5**	**620**

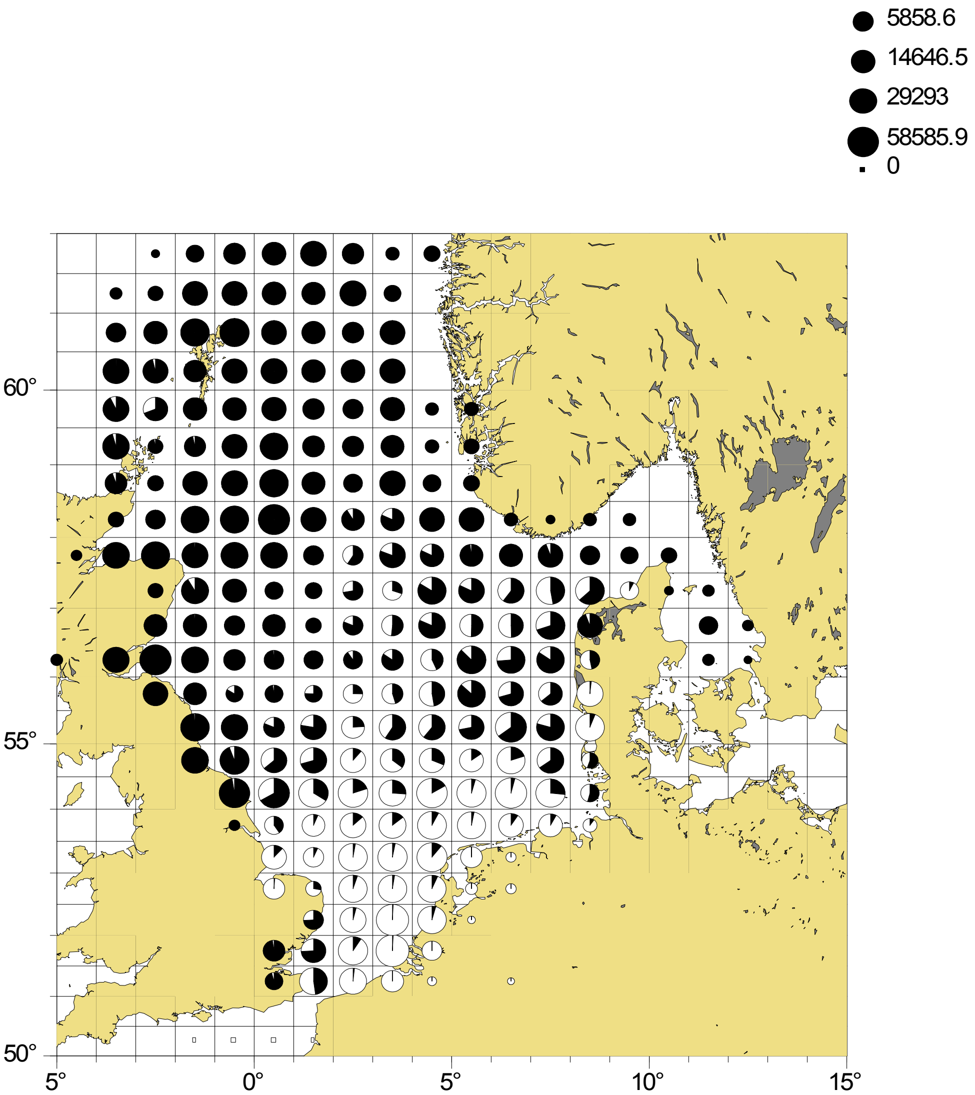
These data underestimate the actual extent of the Dutch beam trawl fleet as there are 22 other complete ICES rectangles south of 55 ^{o}N not used in this analysis, although they support a lower level of beam trawl effort (Rijnsdorp *et al.,* 1998).

4.2.3 Comparing the role of dredging and fishing activity on the dynamics of the marine ecosystem

As stated in the introduction, we have interpreted this in terms of changes in mortality and production of benthic organisms as a tractable way of describing change. We have begun by preparing comparable indices of the spatial extent of dredging and beam trawling. Data available to WGECO suggest that an estimated 284 km^2 of the seabed in the southern North Sea is dredged each year. Despite detailed monitoring systems in UK waters which provide clear evidence that dredging impact is not uniform within a licence area, it is still difficult to describe the precise nature of the impact caused by dredging. It is likely that some areas will be heavily impacted by several tens of hours of dredging in a 100 m × 100 m area which lowers the seabed by up to 10 m, while other parts of the licence area may only experience a single pass of a trailer dredger. Experimental estimates of invertebrate mortality rates resulting from these impacts also suffer from this lack of detailed information as it is difficult to target benthic sampling techniques to precise locations of known impact. Thus, estimates of mortality from experimental studies quoted in Section 4.5.1, citing reductions in abundance of 72 % and 94 %, must be assumed to relate to a uniformly dredged seabed of average intensity.

Similar problems exist in the interpretation of beam trawl spatial distribution and impact. Estimates based on the micro-distribution of the Dutch beam trawl fleet suggest that 4,160 km^2 are trawled more than five times per year, and 620 km^2 are trawled more than ten times per year. This underestimates the total impact by trawling in the southern North Sea as other fleets, particularly the UK and Belgian fleets, operate there, and there is also fishing by other towed gears such as otter trawls (Figure 4.2.3.1.). It can, therefore, be concluded that beam trawling activity is more extensive than dredging activity.

Figure 4.2.3.1 Distribution of bottom trawl (black) and beam trawl (white) effort (hours per year) for 1998 (data from Greenstreet, pers. comm., and Zuhlke *et al.,* 2001).



Data reported in Lindeboom and de Groot (1998) suggested that a single pass of a beam trawl resulted in invertebrate mortality of 7–33 %, but there was considerable uncertainty caused by the potential variability in the distribution of the benthic fauna. Without further data describing the relationship between fishing intensity and increased mortality rates, it is difficult to extrapolate these data further. However, given the direct mortality rates that can be attributed to beam trawling (Figure 4.2.3.2), it is not unreasonable to expect that, for many size classes of benthic fauna, ten or more passes of a beam trawl per year results in high mortality rates that may be comparable to the effect of average levels of dredging activity reported above.

This proposed approach for assessing the impacts of trawling and dredging on mortality could also be extended to assess the effects on production, since relationships between trawling disturbance and the production of macrobenthic fauna can be established empirically (e.g., Figure. 4.2.3.3).

Figure 4.2.3.2. Direct mortality of hard-bodied (A) and soft-bodied (B) organisms of different body sizes following a single pass of a 12-m beam trawl. Data compiled from Lindeboom and de Groot (1988), Bergman and van Santbrink (2000a, 2000b) and unpublished data. From Duplisea *et al* (2001a).

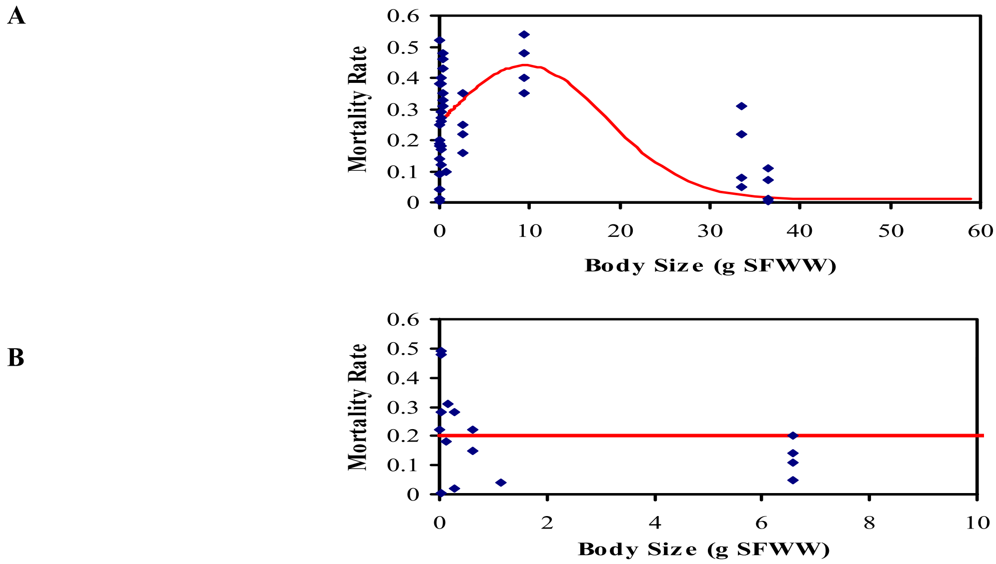
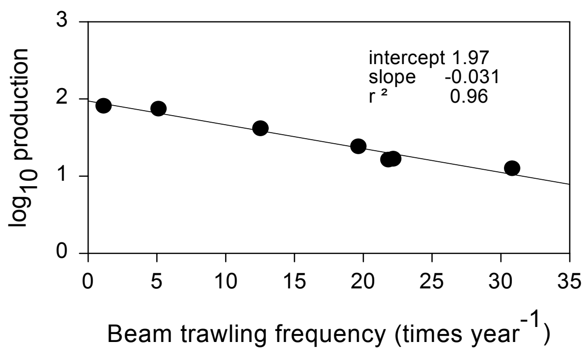


Figure 4.2.3.3. Relationship between benthic infaunal production and beam trawling frequency on a muddy-sand sediment. Source: Jennings *et al.* (2001).



4.2.4 Comparing the role of dredging and fishing activity on nutrient turnover

There have been very few studies of trawling impacts on production or biogeochemistry on the types of sediment that are dredged, so it is inappropriate to compare the effects of these activities at the present time. However, examples provided below show that existing methodologies and protocols could be used to conduct such assessments. Most impact studies which link tow frequency to production or fluxes relate to beam trawls, and more work is needed to provide data that describe the impacts of dredging and other activities on nutrient fluxes.

The relationships between nutrient fluxes and frequency of trawling disturbance can be estimated from empirical data or models although, as with the estimates between production and trawling disturbance, the data are for much finer sediments than those which would be extracted by dredgers, and it is not appropriate to apply them at the present time. Duplisea *et al.* (2001b) modelled the effects of trawling disturbance on nutrient fluxes and chemical concentrations, and models of this type could, with further validation, be used to predict the large-scale effects of trawling. Similarly, Percival and Frid (2000) provide empirical estimates of trawling effects on nutrient fluxes, and their approach could be used as a basis for linking levels of disturbance by beam trawlers or dredges to nutrient fluxes.

4.2.5 Impact of spoil disposal on the dynamics of the marine ecosystem

In very general terms the impact of dumping can be considered equivalent to that of dredging, in that there is a limited focus of intense impact, and a larger surrounding region where disturbance is limited and impacts on benthic invertebrates are minor. In terms of the quantities of material disposed of in the North Sea, approximately 88 mt of dredge spoil was dumped in the North Sea in 1996, approximately double the total amount for the quantity of marine aggregate removed (OSPAR, 2000). The difficulties of describing the impact of sediment disposal will be similar to those described in previous sections, and without specific impact-specific mortality rates these issues cannot be pursued further here. However, the framework laid out in Section 4.2.3 is equally appropriate to this activity.

4.2.6 Conclusion

It is evident that, for a similar level of assumed benthic mortality, the extent of impact of beam trawlers is greater than that of dredgers. There are, however, a number of other factors that must be considered before we can reach firm conclusions about the relative impact of dredging and beam trawling on ecosystem dynamics:

1) The approach to an initial evaluation that we have described has not taken into account any of the secondary impacts of dredging or fishing, particularly the impact of sediment plumes on benthic habitats in surrounding areas.
2) This exercise has shown that we need data on the impacts of beam trawling and dredging on ecosystem dynamics and nutrient turnover that have been collected in comparable habitats.
3) Beam trawling has a chronic impact on invertebrate fauna, and the long-term consequences on the environment of repeated, low level mortality rates will be inherently different from the high levels of instantaneous mortalities suffered at dredge sites. It may also be the case that in homogeneous sandy environments, the effect of dredging activity could also be considered chronic.
4) Our short-term approach to the comparisons between beam trawling and dredging activity does not consider sustainability of the exploited resource. If the spatial extent of the gravel biotope in the southern North Sea is relatively restricted, and if it is assumed that it is a non-renewable resource, then the impact of physical removal of gravel may be significant in the long-term.
5) A long-term assessment of the impact of a number of different human activities on the marine environment must take into account recovery rates of benthic organisms. There are insufficient data to compare the recovery of benthic biomass and production following impacts on the same habitat types.
6) The extent of the habitat types impacted by human activities is required to evaluate the broader-scale impact of site-specific mortality. The detailed spatial extent of the gravel and sand biotopes in the southern North Sea is poorly known, and without these data it is not possible to provide habitat-specific interpretations of the impacts of each activity.
7) Ideally it is necessary to evaluate the impact of all activities on the marine environment, and consider whether, in areas where individual impacts were low, there may be in-combination effects where several human activities occur together.

References

Bergman, M.J.N., and van Santbrink, J.W. 1997. Verspreiding en abundantie van grotere (epi)bentische macrofaunasoorten op het NCP en de invloed van boomkorvisserij op de verspreiding van deze fauna. BEON data report/NIOZ 96 V.

Bergman, M J N, and van Santbrink, J.W. 2000a. Mortality in megafaunal benthic populations caused by trawl fisheries on the Dutch continental shelf in the North Sea in 1994. ICES Journal of Marine Science, 57: 1321-1331

Bergman, M.J.N, and van Santbrink, J.W. 2000b. Fishing mortality of populations of megafauna in sandy sediments. *In* Effects of fishing on non-target species and habitats: biological, conservation and socio-economic issues. Ed. by Kaiser, M.J., and de Groot, S.J. Oxford, Blackwell Science.

Boyd, S. E., and Rees, H. L. 2001. Effects of dredging intensity on the macrobenthos in commercial aggregate extraction sites in the English Channel. Marine Pollution Bulletin (in press).

Collie, J.S., Hall, S.J., Kaiser, M.J., and Poiner, I.R. 2000. A quantitative analysis of fishing impacts on shelf sea benthos. Journal of Animal Ecology, 69: 785–798.

Crown Estate. 2001. Marine aggregate dredging: The area involved – 3rd annual report. The Crown Estate, 16 Carlton House Terrace, London, SW1Y 5AH, UK.

Desprez, M. 2000. Sedimentological and biological effects of sand deposition around the dredging site of Dieppe (F). ICES CM 2000/E:07. 8pp.

Desprez, M., and Duhamel, S. 1993. Comparison of the impact of gravel extraction on geomorphology,sediment & macrofauna in two areas: Klaverbank (NL) and Dieppe (F). ICES CM1993/E:7, pp.51–66.

Duplisea, D. E., Warr, K. J., Dinmore, T. A., and Jennings, S. 2001a. A size-based model to predict the impacts of bottom trawling on benthic community structure. Final Seedcorn Project Report to CEFAS, Lowestoft, UK

Duplisea, D.E., Jennings, S., Malcolm, S.J., Parker, R., and Sivyer, D. 2001b. Modelling the potential impacts of bottom trawl fisheries on soft sediment biochemistry in the North Sea. Geochemical Transactions, 14: 1-6.

ICES. 1992a. Report of the Study Group on Ecosystem Effects of Fishing Activities. ICES CM 1992/G:11.

ICES. 1992b. Report of the ICES Working Group on the Effects of Extraction of Marine Sediments on Fisheries. ICES Cooperative Research Report No. 182. 78pp.

ICES. 2000. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 2000/ACME:02.

ICES. 2001, Report of the Working Group on the Effects of Extraction of Marine Sediments on the Marine Ecosystem. ICES CM 2001/E:06.

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

Kenny, A. J., and Rees, H. L. 1994. The effects of marine gravel extraction on the macrobenthos: early post dredging recolonisation. Marine Pollution Bulletin, 28(7): 1–19.

Kenny, A. J. and Rees, H. L. 1996. The effects of marine gravel extraction on the macrobenthos: Results 2 years post-dredging. Marine Pollution Bulletin, 32(8/9): 615–622.

Kenny, A. J., Rees, H. L., Greening, J., and Campbell, S. 1998. The effects of gravel extraction on the macrobenthos at an experimental dredge site off North Norfolk, UK (results 3 years post-dredging). ICES CM1998/V:14, 1–7.

Lees, G., Kenny, A. J., and Pearson, R. 1992. The condition of benthic fauna in suction trailer dredger outwash: initial findings. ICES CM.1992/E:7, 22–28.

Lindeboom, H.J., and de Groot, S.J., 1998. The effects of different types of fisheries on the North Sea and Irish Sea benthic ecosystems. Netherlands Institute of Sea Research, Texel.

Moran, P.J. 1991. The effects of dredging on the larval settlement and community development of fouling organisms on Port Kembla harbour, Australia. Water Research, 25: 1151–1155.

Newell, R. C., Seiderer, L. J., and Hitchcock, D. R. 1998. The impact of dredging works in coastal waters: A review of the sensitivity to disturbance and subsequent recovery of biological resources on the sea bed.

Norden Andersen, O. G. N., Nielsen, P. E., and Leth, J. 1992. Effects of sea bed, benthic fauna and hydrography of sand dredging in Køge Bay, Denmark. 1-6. Helsingor, Denmark. Proceedings of the 12th Baltic Marine Biologists Symposium.

OSPAR. 2000. Quality Status Report Region II, Greater North Sea. OSPAR Commission, London. 129 pp.

Percival, P., and Frid, C.L.J. 2000. The impact of fishing disturbance on benthic nutrient regeneration and flux rate. ICES CM 2000/ Z:07.

Resource Analysis. 1998. Prioritisation of human pressures for the Greater North Sea Quality Status Report 1999, an interactive and iterative Multi Criteria Analysis Approach. Results and Methodology. Delft, Report RA/98-332. December 1998.

Rijnsdorp, A.D., Buys, A.M., Storbeck, F., and Visser, E.G. 1998. Micro-scale distribution of beam trawl effort in the southern North Sea between 1993 and 1996 in relation to the trawling frequency of the sea bed and the impact on benthic organisms. ICES Journal of Marine Science, 55: 403–419.

van Moorsel, G.W.N.M. 1993. Long term recovery of geomorphology and population development of large molluscs after gravel extraction at the Klaverbank (North Sea). 92.16, 1–41.

van Moorsel, G. W.N.M. 1994. The Klaverbank (North Sea), geomorphology, macrobenthic ecology and the effect of gravel extraction. 94.24, 1-65. Bureau Waardenburg bv, Culemborg, the Netherlands.

Zuhlke, R., Alvsvag, J., de Boois, I., Cotter, J., Ford, A., Hinz, H., Lancaster, J., Piet, G., and Prince, P. 2001. Epibenthic diversity in the North Sea. Report to EC.

5 TESTING HYPOTHESES ABOUT THE SENSITIVITY OF ECOSYSTEM COMPONENTS TO BOTTOM FISHING IMPACTS

ToR c: *"continue the work plan to test hypotheses about which components of the marine ecosystem are most sensitive to bottom fishing impacts."*

5.1 Introduction

This work is a continuation of the testing of metrics of ecosystem quality developed at previous meetings of WGEGO. At the 2001 meeting (ICES, 2001), WGECO examined the sensitivity of demersal fish communities to fishing, based on trends in mean life history characteristics for the community. Within populations, selective fishing mortality is expected to lead to increases in growth rate and reductions in age and size at maturity. Within communities, selective mortality leads to reduced abundance of large species with low intrinsic rates of increase, and dominance of smaller species with higher intrinsic rates of increase. Variation in life history characteristics within populations is much lower than among species in a community, and thus selective effects of fishing are most readily observed at the community level. Previous analyses have confirmed the trends in the structure of fish communities that were expected from life history theory.

Changes in size distributions in response to exploitation have also been described. As fishing mortality increases, mean size of individuals in the community drops, and large individuals form a smaller proportion of the biomass. Consequently, the (negative) slope of size spectra generally became steeper while the intercept increased. These effects have been demonstrated in fisheries ranging from the tropics to the Arctic (e.g., Gislason, 1994; Hall, 1999; Gislason and Sinclair, 2000; Rogers and Ellis, 2000). Size-based approaches such as these provide an effective way of describing gross community responses to fishing, but the structure of the size spectrum and the observed response is based on a combination of factors including: (1) differential vulnerability of larger species; (2) within-population changes in mean size; (3) genetic changes in life history; and (4) predator-prey relationships within the community. Although it is difficult to attribute observed changes in size spectra to a specific factor, existing theoretical models explain the structure of size spectra in response to factors 1, 2, and 4.

Potential weaknesses of previous analyses were (1) lack of reference to quantitative differences in fishing effort when making temporal and spatial comparisons among fish communities; (2) limited consideration of differential responses of specific species groups in their contribution to patterns of change in the community; and (3) lack of explicit analyses of the effects of sampling gears on the properties of size-spectra and life-history based metrics. Moreover, we did not assess whether the trophic structure of the community was sensitive to fishing impacts.

The following analyses test hypotheses about the components of the marine ecosystem that are most sensitive to bottom fishing impacts by comparing different responses of marine communities to variations in fishing effort. For a series of geographic regions, available data are used to compare the sensitivity of the fish community to fishing impacts, based on analyses of diversity, trophic level, size-spectra and other size-based metrics. Specifically, we compare the responses of the selected metrics to fishing when they are (1) based on samples collected in the same areas over similar time periods with different sampling gears; (2) used to compare areas which are subject to different degrees of fishing intensity; and

(3) used to compare different ecosystems. Not all metrics can be applied to all ecosystems, because appropriate data were not always available.

The work using trophic metrics is new and thus requires specific background information. Larger individuals in the trophic continuum feed at higher trophic levels, a pattern demonstrated empirically for plankton, benthic invertebrate and fish communities (Fry and Quinones, 1994; France *et al.,* 1998; Jennings *et al.,* 2001). Since fishing intensity is positively correlated with the slopes of size spectra (Rice and Gislason, 1996; Bianchi *et al.,* 2000), we would expect changes in the slope of the size spectrum to be associated with changes in the mean trophic level of the community.

Changes in the trophic structure of landings and fish communities due to fishing have been described using diet data (Yang, 1982), Ecopath estimates of trophic level (Pauly *et al.,* 1998), and more recently, species-specific estimates of trophic level from nitrogen stable isotope analysis (Pinnegar *et al.,* 2002). Observed decreases result from changes in species composition because there are weak cross relationships between the trophic level and body size (vulnerability) or commercial value (desirability) across species (Pinnegar *et al.,* 2002). However, the existing analyses of fishing impacts on trophic level have considered the effects of changes in species rather than size composition, and assumed that the trophic level of species or species-groups was fixed. This approach overlooks the significant impact of fishing on the size structure of populations (Beverton and Holt, 1957) as well as changes in trophic level with body size (Jennings *et al.,* 2002). For example, many species will switch from plankton feeders to piscivores as they grow in mass by 4–5 orders of magnitude (Cushing, 1975). It is also conceivable that trophic level decreases with body size. Some flatfishes, for example, shift from feeding on predatory polychaetes to deposit or filter-feeding bivalves as they grow (Braber and de Groot, 1973). The effect of intraspecific changes in trophic level is that the real trophic response of an exploited community is likely to differ from that predicted when trophic level is not treated as a function of body size.

The main impediment to the quantification of relationships between body size and trophic level is the absence of size-related trophic level estimates for many species. Diet data have been used to estimate the trophic levels of the main North Sea fish species and, for a few species, to examine relationships between body size or age and diet (Yang, 1982; Christensen, 1995; Greenstreet, 1996). However, short-term dietary data may not provide a good assessment of the trophic level of species that switch diet frequently, prey on species that are digested at different rates and have gut contents that cannot be identified.

An empirical (rather than model-based) approach was used to assess whether the trophic structure of the community was sensitive to fishing impacts. We define trophic level operationally as the biomass-weighted mean trophic level of species captured in a particular survey. Since sampling gear is both size and habitat selective, mean trophic level derived from survey data at best provides an index of community trophic level. At worst, estimates of trend could be biased, particularly if there have been shifts in dominance between benthic and pelagic communities.

5.2 Available data sets

Data were available at the meeting from four different ecosystems and included seven sets of data from different research surveys. The ecosystems are the North Sea, Scotian Shelf, Barents Sea, and Atlantic off the Portuguese coast. The survey data include 4 sets from the North Sea: Scottish Groundfish Survey (SAGFS), International Bottom Trawl Survey (IBTS), Sole Net Survey (SNS) and Beam Trawl Survey (BTS). There was also one set of survey data from each of the Scotian Shelf, Barents Sea and from Portugal. Data on fishing effort were available for some of these regions. The analyses carried out on each data set varied according to the type of data available, but as far as possible similar analyses were carried out in all areas.

5.2.1 Ecosystem descriptions

There are large differences between the four systems, both in terms of hydrography and in terms of the history of fishing. These differences may be important in determining how each system has behaved in response to different levels of fishing effort. A short description of their main characteristics is given below.

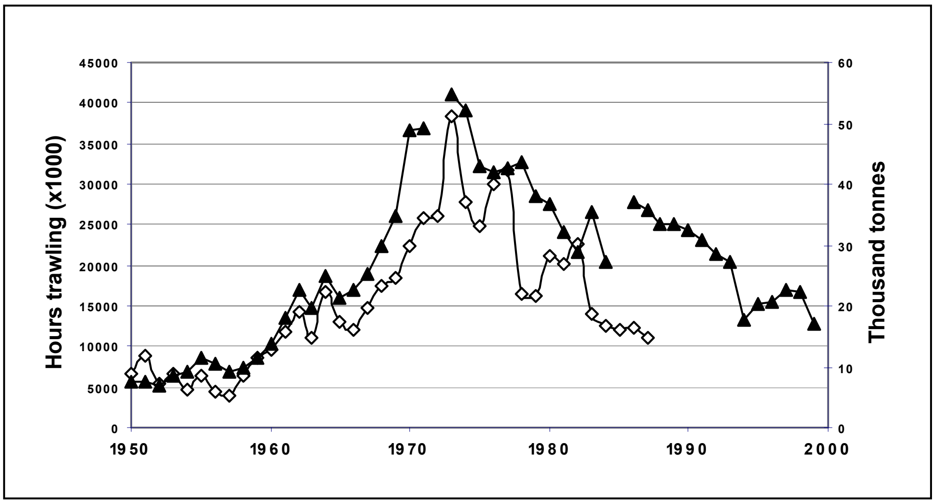
The North Sea is a shallow semi-enclosed sea area that is well documented. Fishing has a long history here and landings remained at a similar level from the early 20th century until the mid-1960s when increased landings of gadoids and industrial fishing increased the landings considerably. However, since then, landings have been declining.

The Scotian Shelf is a well-defined underwater geographic unit. It is a cold, boreal system, with complex water circulation characteristics because of the topography (including banks and deep basins) and is influenced by different currents. The southwest is under the influence of a relatively warm regime, while the northeast part of the shelf holds colder waters. For most species these two areas are managed separately. The more productive western area has been highly exploited since the 1960s. The principle ground fisheries of the eastern Scotian Shelf were overexploited in the late 1980s/early 1990s, and a fishery closure was imposed in 1993, which is still in place.

The Barents Sea is a frontal system, receiving water from the Atlantic in the Gulf Stream from the south and Arctic waters from the north. It is generally shallow, with a prevailing depth of between 100 m to 300 m. The Barents Sea has a rather complicated trophic structure as the available prey species vary according to the migrations of both pelagic and demersal species. The most important prey species are capelin and polar cod from Arctic waters and herring from boreal waters. It is unclear what influence the demersal fishery has on the community make up, but it should be noted that this fishery uses a large mesh size (≥ 125 mm) and for the recent years or so technical conservation measures have been introduced. Furthermore within the Barents Sea there are areas that are not fished. These are unsuitable for bottom trawling and are closed to gears that target pelagic species as a conservation measure to protect 0-group capelin.

The Atlantic off the Portuguese coast has a narrow continental shelf, and is strongly influenced by the two main oceanographic regimes in this area, which change seasonally. During the summer, from June to October, the northern extreme of the Canary upwelling system reaches Portugal and brings colder and nutrient-rich waters to the surface. In the winter there is coastal convergence and warmer, offshore waters reach the shelf. Furthermore, the presence of denser, more saline Mediterranean waters brings a number of Mediterranean and subtropical species to the region. These environmental factors influence the trophic structure of the system, for example, upwelling influences the recruitment of pelagic (prey) species. Fishing is a traditional activity in Portugal, though from Figure 5.2.1.1, it can be seen that over recent years both effort and landings have been steadily decreasing. This, combined with the results of both last year's and this year's analyses, suggests that in this region factors other than fishing are shaping the fish assemblages.

Figure 5.2.1.1. Landings (black triangles ▲) and effort (white squares ◇) by the Portuguese trawl fleet for the period 1950 to 1999.



5.2.2 North Sea surveys

Scottish August Groundfish Surveys

Groundfish survey data collected during the Scottish August Groundfish Surveys were available for 75 ICES statistical rectangles in the northwestern North Sea for a period from 1983 to 1996. These surveys were all carried out with the FRV "Scotia (II)", using a 48-foot Aberdeen Otter Trawl, towed for one hour (Greenstreet and Hall, 1996; Greenstreet *et al.,* 1999a).

International Bottom Trawl Survey (IBTS)

The International Bottom Trawl Survey (IBTS) is a follow-up of the International Young Fish Survey (IYFS) that was conducted in the North Sea and Skagerrak/Kattegat in February of each year starting in the late 1960s. Over the years the survey has changed from a survey on young herring into one for demersal fish and herring of all ages and sizes. At the same time, the area surveyed has expanded until from 1974 onwards the whole North Sea proper, Skagerrak and Kattegat were covered. The IBTS was conducted in international collaboration with different research vessels covering specific areas. Over time standardization in gear type, rigging specifications and sampling strategy was pursued among participating countries. During the early years of the survey, a 78-foot Dutch herring trawl was recommended as the standard gear but in 1977 it was decided to use the GOV-trawl (Grande Ouverture Verticale) as standard gear. From then onward most vessels used GOV but it took several years before it was adopted by all vessels. For consistency, only the years from 1980 onwards conducted in the first quarter using GOV were used (Figure 5.2.2.1). The GOV has a high vertical net opening of 5 m to 6 m. The horizontal opening of the net is approximately 20 m. Standard fishing speed is 4 knots measured as trawl speed over the ground. Each haul lasts 30 minutes.

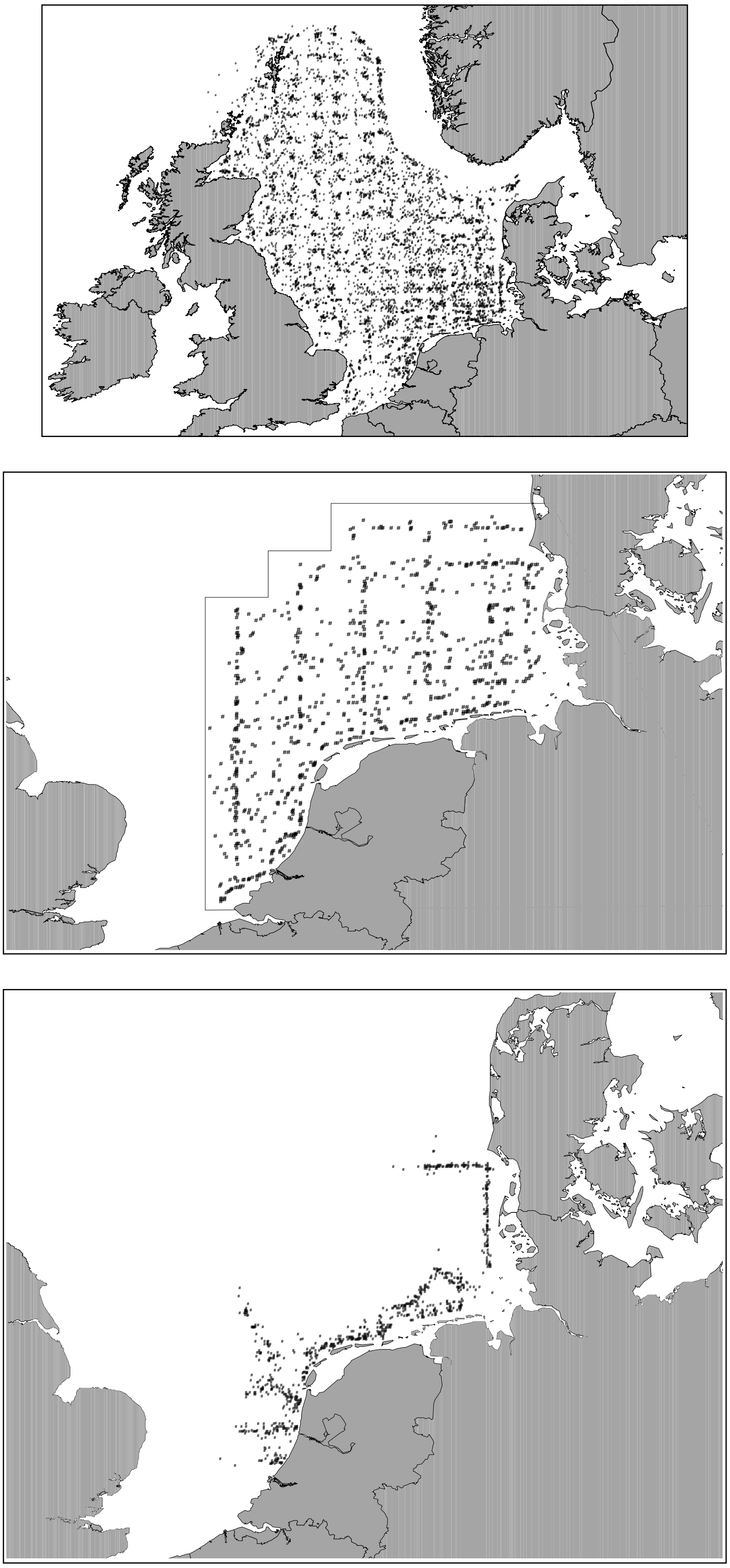
Beam Trawl Survey (BTS)

The BTS survey was initiated in 1985 and aims at obtaining abundance estimates of the dominant age groups of plaice and sole including pre-recruits. The survey is conducted in the third quarter. The fishing gear used is a pair of 8-m beam trawls rigged with nets of 120 mm and 80 mm stretched mesh in the body and 40 mm stretched mesh cod-ends. A total of 8 tickler chains are used, 4 mounted between the shoes and 4 from the groundrope. The survey was designed to take between one and three hauls per ICES rectangle depending on the rectangle (Figure 5.2.2.1). The stations are allocated over the fishable area of the rectangle on a “pseudo-random” basis to ensure that there is a reasonable spread within each rectangle. No attempt is made to return to the same tow positions each year. Towing speed is 4 knots for a tow duration of 30 minutes and fishing occurs during daylight only. From the start of BTS in 1985 until present the same research vessel (RV “Isis”) has been used.

Sole Net Survey (SNS)

This survey started in 1969 and was carried out using the RV “Tridens” until 1995 and RV “Isis” from 1996 (Figure 5.2.2.1). The survey is conducted in the third quarter. The gear used is a pair of heavy 6-m beam trawls with 40-mm stretched mesh cod-ends and 4 tickler chains. Fishing speed is 3.5–4 knots and duration of a haul is 15 minutes. The survey fishes a series of fixed transects with stations perpendicular to the coast. All flatfish are sampled for length and age and other species for length only. Indices for age groups 0–3 have been prepared annually from 1977 as numbers per 100 h fishing averaged from all stations covered.

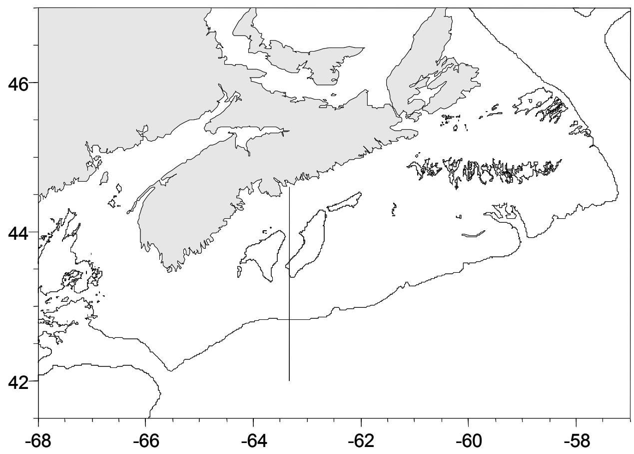
Figure 5.2.2.1. Positions of hauls in three surveys in the North Sea. From top to bottom: IBTS, BTS and SNS.



5.2.3 Scotian Shelf

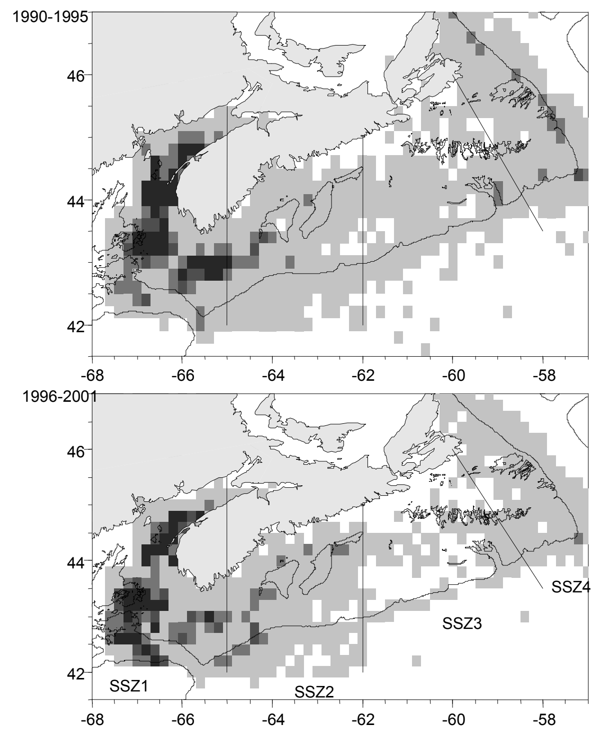
Data were extracted from the summer RV survey series from the Scotian Shelf into two subsets. The first was the location, time, species code of the catch in terms of numbers and biomass for the 30-year period 1970–2001. There were about 57,000 observations of 102 species. The second set was all observed lengths from the same RV survey, 340,000 entries of 56 species.

Figure 5.2.3.1. The Scotian Shelf. The depth contour shown is 200 metres. The line at 63.33 °W divides the Shelf into eastern and western portions. The verticle line marks the boundary between the east and west Scotian Shelf.



A third data set was also available of all the trawler effort having location data which covered 1990–2000. These data were aggregated to 0.1 degree squares. The units are days fishing, which although crude in terms of effort had the most coverage of geographically defined data. The relative intensity of effort over the periods 1990–1995 and 1996–2001 are shown in Figure 5.2.3.2. The totals in each of the zones for the 11-year period are 430, 71, 10, and 42 thousand days, respectively.

Figure 5.2.3.2. Relative trawler effort (days fished) distribution on the Scotian Shelf from 1990-1995 and 1996-2001. The three straight lines divide the effort data into four zones (SSZ1-4) used in subsequent analysis.



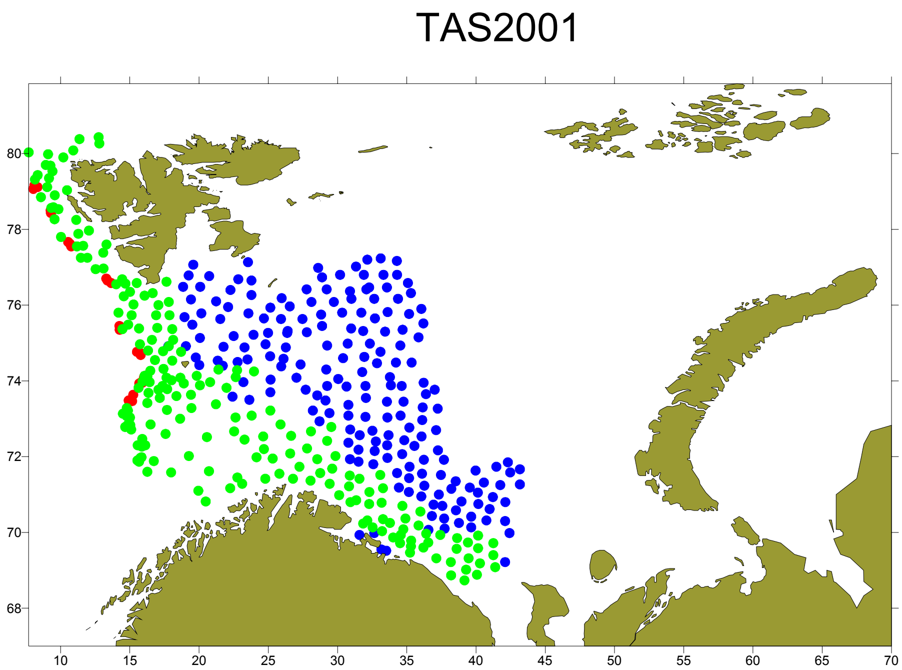
5.2.4 Barents Sea

The Barents Sea was considered as the area delimited by 82–84 ºN lat. in the north; by the Murman and Norwegian coasts in the south (to 67 ºN lat.); by Novaya Zemlya and Franz Josef Land in the east; and by the continental slope (to approx. 800–1000 m depth) in the west.

The effects of fishing on the characteristics of the demersal fish community were tested using data from Russian bottom trawl surveys. These surveys are conducted by the Polar Institute (PINRO) from October to December of each year beginning in the early 1980s. Since 1996, data on biology and size distribution of non-target fish have been collected. Trawl stations extend from northern Spitsbergen to the Norwegian coast and Novaya Zemlya (Figure 5.2.4.1).

A research bottom trawl was used as the standard sampling gear. The trawl had a vertical net opening of 8 m, and a horizontal opening of approximately 20 m. Standard fishing speed was 3.5 knots. Hauls were 1 hour in duration.

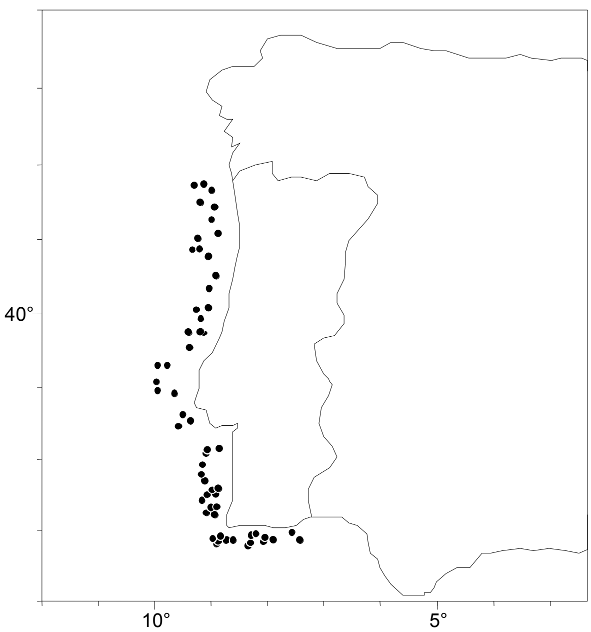
Figure 5.2.4.1. Distribution of trawl stations in the Barents Sea during the Russian bottom trawl survey in 2001.



5.2.5 Atlantic off the Portuguese coast

Data are from the Portuguese autumn bottom trawl research surveys (Figure 5.2.5.1). These surveys are described in Cardador *et al.* (1997). Data on the total abundance of fish are available for 1982, 1985, 1987, and 1989 to 2001. Length frequency data are available for 1990 onwards.

Figure 5.2.5.1. Map of research survey stations sampled during the 2001 Portuguese autumn survey.



5.3 Analyses on a region-by-region basis

5.3.1 The sensitivity of the northwestern North Sea fish communities to bottom fishing impacts

5.3.1.1 Introduction

Past work of WGECO and others (e.g., Greenstreet and Hall, 1996; Greenstreet *et al.,* 1999a; Jennings *et al.*, 1998; Jennings *et al.*, in prep; Jennings *et al.,* 1999a) has led to the proposal of three *a priori* hypotheses as to how some characteristics of the groundfish assemblage are affected by fishing:

1) The species richness and species diversity of the groundfish assemblage should be lower in areas most disturbed by fishing;

2) The life-history characteristics of the groundfish assemblage should change, growth rates should be highest, and size at maturity, age at maturity, and ultimate body size should be lowest, in areas most disturbed by fishing;

3) The trophic level at which fish belonging to the groundfish assemblage feed should be lower in areas most disturbed by fishing.

All three predictions follow from first-order effects of fishing as a source of mortality that is not equal across all species and sizes of fish in the community. More complex ecological processes, such as inter-specific competition, top-down/bottom-up control, food supply, trophic level transfer efficiency, and productivity (Connell, 1975, 1978; Paine, 1974; Huston, 1994; Pauly and Christensen, 1995) could all serve to amplify these changes (see Text Box 1, Section 5.4.).

We analysed Scottish August Groundfish Survey (SAGFS) data, international and Scottish fishing effort data, and information on life-history characteristics and trophic level of the species encountered in the SAGFS, to examine how fishing affects these community characteristics. We combined spatial and temporal analyses in an attempt to strengthen the case that the changes observed were in fact caused by fishing.

5.3.1.2 Analytical design

We examined data for 75 ICES rectangles, divided initially into three groups, or treatments, of low, medium and high "current" fishing disturbance, and tested the hypothesis that each characteristic of the demersal fish community is "most affected" in the rectangles of highest disturbance, and least affected in the rectangles of lowest disturbance. Systematic changes to population abundance and population age structure occur when mortality rates change. Fish communities may therefore be most affected by the rate of change in levels of disturbance, rather than the actual level of disturbance experienced. To investigate this, we redivided the rectangles into a second set of three treatments: one group in which fishing disturbance levels have declined over recent decades, a second where fishing disturbance levels have increased slowly, and the third where fishing has increased rapidly. We then tested the hypotheses that each community metric is either unchanged, or perhaps "improved", in rectangles where disturbance has declined, and most perturbed in the rectangles where fishing disturbance levels have increased fastest.

Even if these first hypotheses are supported by the data, this does not confirm that fishing has caused the changes. Current fishing levels may be highest, or rates of change in fishing disturbance may have increased most quickly in rectangles where species richness and diversity are highest, where fish growth rates are quickest, ultimate body size and length at maturity are smallest, and age at maturity is lowest, or where fish in the assemblage feed at a lower trophic level. To discount this alternative interpretation, we examined long-term time series trends for two sets of sub-groups of rectangles. If fishing has caused the change in the community characteristics, then predictable temporal trends should be apparent. Little or no long-term trend should be apparent in rectangles where fishing disturbance is low or relatively unchanged in recent decades, whereas in rectangles affected by fishing, temporal trends in a predictable trend should be detected. The greater the impact from fishing, the steeper the gradient should be. An assumption underlying this analytical design is that prior to fishing impact, the community characteristics in the two sets of treatments had the same start point.

In adopting this analytical design, we have attempted to follow, as far as possible, a one-way ANOVA design. However, it is important to realise that the distribution of fishing effort was not random across the 75 rectangles (Jennings *et al.*, 1999b; Greenstreet *et al.*, 1999b). A true ANOVA design would have had the random distribution of the "treatment" across the 75 rectangles. This has two major implications:

- Spatial variation could introduce a potentially confounding effect. Concentration of the impact of fishing into restricted areas could magnify the effect of fishing on the demersal fish community. Nevertheless, this is still a fishing effect. It will lead to similar distribution in the community characteristic being investigated. The question is, can this spatial factor introduce the sort of trends we anticipate independently of fishing?

- The spatial cohesion of both the "treatment" and the "effect" could, through spatial auto-correlation, reduce the independence of the data. This has consequences with respect to estimation of the actual degrees of freedom in any statistical analysis. While we have presented significance levels for the ANOVA results, some caution is necessary in interpreting them. We have for this reason adopted a significance level of $P<0.01$ in order to reduce the chances of rejecting the null hypothesis and incorrectly inferring an effect of fishing. Analysis of groundfish survey data collected at high spatial resolution (25 to 30 half-hour GOV samples collected within a 20 km by 20 km area) suggests that auto-correlation between species abundance is almost entirely diminished at distances of around 10 km to 15 km. Examination of variograms for each of the community characteristics indicates that spatial auto-correlation is diminished over a distance of around 150 km (two to three ICES rectangles), suggesting that the true number of degrees of freedom may be as few as 25 to 40.

5.3.1.3 The data sets

5.3.1.3.1 Groundfish survey data

Scottish August Groundfish Survey (SAGFS) data were examined from 75 ICES statistical rectangles located in the northwestern North Sea where survey coverage was most complete (Figure 5.3.1.3.1.1). Only trawl samples collected using a 48 foot Aberdeen Otter Trawl towed for one hour were included in the data set. Data for those groundfish species likely to be well sampled by the gear were analysed. Pelagic species and other species not well sampled by the 48-ft Aberdeen otter trawl, such as herring, sprats and sandeels, were all excluded. The results therefore only apply to the demersal groundfish community occupying the area. For more details regarding the data, see Greenstreet and Hall (1996) and Greenstreet *et al.* (1999a).

To determine "contemporary" levels of each of the community metrics to be examined, trawl species abundance data covering a period of 14 years from 1983 to 1996 were extracted from the SAGFS database. All the trawl samples were collected by the same survey vessel, FRV "Scotia (II)". For one rectangle only ten trawl samples were available. This rectangle was not sampled in 1983, 1985, 1987, or 1995. To avoid sample size dependency problems, sampling effort was standardised to ten trawls in the other 74 rectangles by excluding, as necessary, trawl samples selected at random from these four years. Previous analysis of SAGFS data has indicated that it is necessary to aggregate at least five one-hour trawl samples in order to derive reliable community metrics. All ten trawl samples in each rectangle were therefore combined to provide a single aggregated, highly standardised, species abundance sample for each rectangle upon which to calculate each community metric.

For the second analysis, looking at "long-term temporal trends" in the structure and composition of the groundfish assemblage in rectangles varying in the level of fishing effort to which they had been subjected, data from the full time series, spanning the period 1925 to 1996, were used. Following Greenstreet *et al.* (1999a, 1999b), data were pooled into "groups" of two or three years to ensure adequate sampling effort in each time-period/effort "treatment" cell.

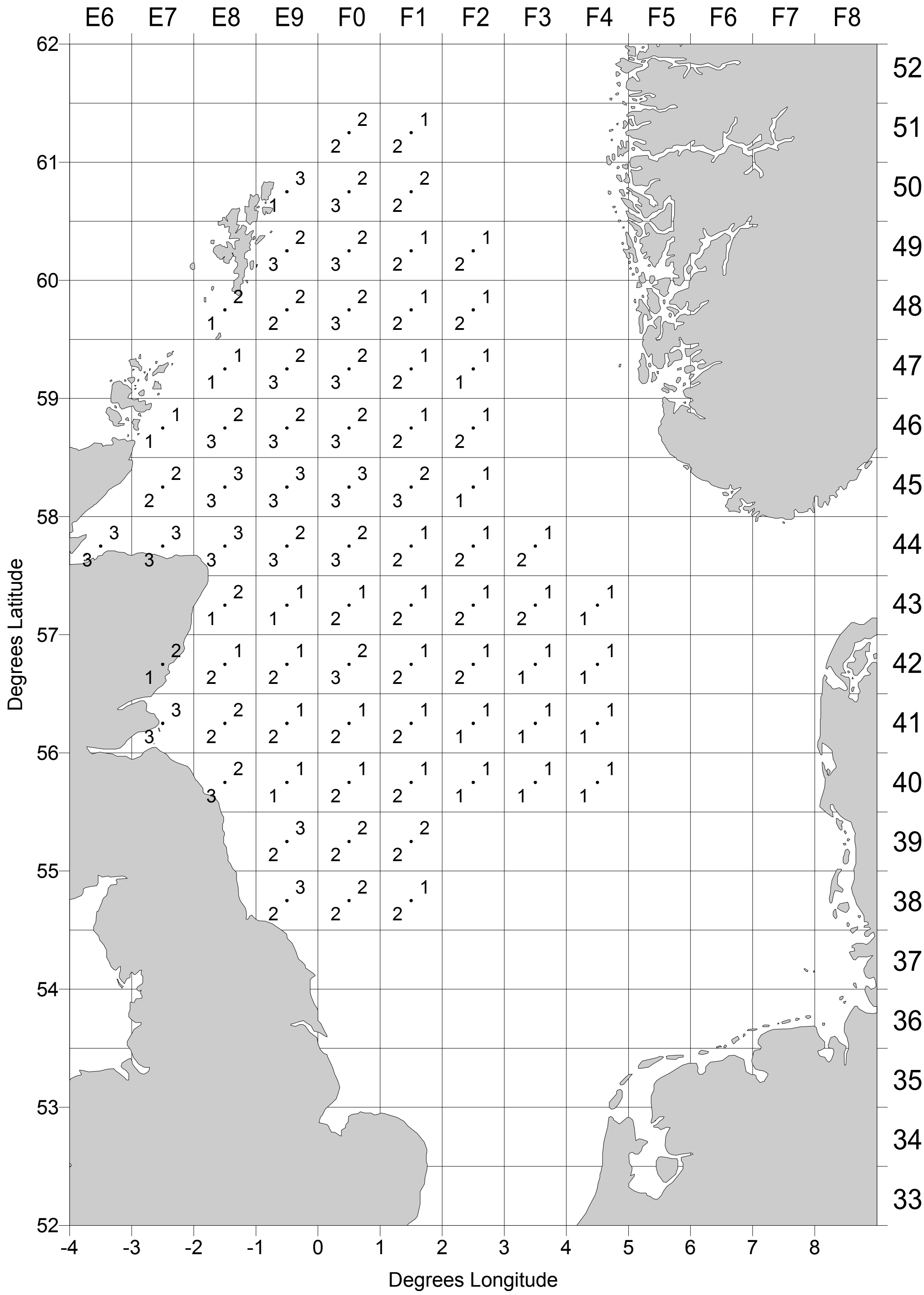
5.3.1.3.2 Fishing effort data

International otter trawl, beam trawl, and Seine net fishing effort (hours fished) for the period 1990 to 1995 were available from the database compiled during a previous CEFAS/EC project (Jennings *et al.*, 1999b, 2000). Average annual effort values were calculated to provide estimates of the "current" spatial distribution of fishing effort across the 75 ICES statistical rectangles for which groundfish survey data were available. Total annual average fishing effort across the 75 statistical rectangles amounted to 963,216 hours of fishing, 67 % of which consisted of otter trawling, 12 % beam trawling, and 21 % Seine netting. Being the predominant method employed during the early 1990s in this part of the North Sea, our attention was directed towards the effect of variation in otter trawl effort on the groundfish community occupying the area. Otter trawl effort ranged from 645 h yr^{-1} to 63,794 h yr^{-1} across the 75 ICES statistical rectangles. Three broad categories ("treatments") were defined: 40 rectangles where otter trawling intensity was low, from 0 to 4999 h y^{-1}, 25 rectangles of medium otter trawl effort, from 5000 to 19,999 h y^{-1}, and 10 rectangles of high otter trawling intensity, exceeding 20,000 h y^{-1}. The distribution of rectangles belonging to each of these treatments is indicated in Figure 5.3.1.3.1.1.

The international fishing effort database, covering only the years 1990 to 1995, was too short to provide "long-term effort trend" information. Scottish vessels landing in Scotland account for most of the fishing effort in this part of the North Sea. Therefore, to examine the effects of "longer-term trends" in fishing effort on community metrics, the Scottish fishing effort database, extending further back in time, was used instead (Greenstreet *et al.*, 1999b). Effort data for otter trawl were available for the period 1970 to 1994 for each rectangle. Average annual effort for the five-year periods 1970 to

1974 and 1990 to 1994 were computed for both gears. The difference between these values was divided by 20 to provide an "annual rate of change" index for each gear in each rectangle for the period 1970 to 1994. Beam trawling is a relatively recent innovation in the northwestern North Sea, and effort data for this gear were only recorded from 1984 onwards so the same approach could not be adopted for this gear. Annual rates of change in otter trawl effort varied from the extreme outlier of –2,268 h y^{-1} (possibly a reporting error) to 991 h y^{-1}. Three groups of rectangles were again defined: a group of 19 rectangles where otter trawl effort was declining; a group of 35 rectangles where effort was increasing slowly, between 0 and 199 h y^{-1}; and a group of 21 rectangles where effort was increasing rapidly, between 200 and 991 h y^{-1}. The distribution of rectangles belonging to each of these treatments is also indicated in Figure 5.3.1.3.1.1.

Figure 5.3.1.3.1.1. Chart of the northwestern North Sea indicating the 75 ICES statistical rectangles for which SAGFS and effort data were analysed. Effort treatment codes area indicated: current effort treatments upper right codes, 1 = Low, 2 = Medium, 3 = High; rate of change in effort treatments lower left, 1 = Decline, 2 = Slow Increase, 3 = Fast Increase.



5.3.1.3.3 Community metrics

Species richness was simply the count of all species encountered in the aggregated samples. Two diversity indices, Hill's (1973) N1 and N2, were also computed for each of the 75 rectangles' aggregated samples. Hill's N1 diversity index is the exponential of the Shannon-Weiner index and N2 is the reciprocal of Simpson's index (Magurran, 1988).

5.3.1.3.4 Species life-history characteristics

Information regarding four life-history characteristics, $Length_{Infinity}$, Growth Rates, $Age_{Maturity}$, and $Length_{Maturity}$, was available for 28 of the 56 species included in the SAGFS database (Table 5.3.1.3.4.1). $Length_{Infinity}$ and Growth Rate were the parameter values determined from the von Bertolanffy growth equation calculated for each species. The von Bertolanffy growth parameter is not strictly a rate value, but is used here as an index equivalent to growth rate. $Age_{Maturity}$ and $Length_{Maturity}$ values were determined by observation, either from recent survey data, or with recourse to the literature (Jennings *et al.*, 1998, 1999a). These 28 species accounted for over 98 % of the individuals sampled by the SAGFS in any of the spatial/temporal "treatments". Species abundance data were converted to the number of individuals with particular characteristic values, and the mean value for each characteristic for each spatial/temporal "treatment" was computed.

Table 5.3.1.3.4.1. List of species for which life-history character information was available.

		Len_{In}	GR	Age_{mat}	Len_{mat}
Lesser spotted dogfish	*Scyliorhinus canicula*	90.00	0.20	5.00	58.00
Spurdog	*Squalus acanthias*	90.20	0.15	6.50	66.80
Spotted ray	*Raja montagui*	97.80	0.15	6.00	66.60
Cuckoo ray	*Raja naevus*	91.64	0.11	9.00	58.82
Starry ray	*Raja radiata*	66.00	0.23	4.00	45.70
Torsk	*Brosme brosme*	88.60	0.08	9.60	49.55
Cod	*Gadus morhua*	123.10	0.23	3.80	69.70
Four-bearded rockling	*Enchelyopus cimbrius*	36.00	0.20	3.00	14.00
Haddock	*Melanogrammus aeglefinus*	68.30	0.19	2.50	33.50
Whiting	*Merlangius merlangus*	42.40	0.32	1.50	20.20
Saithe	*Pollachius virens*	177.10	0.07	4.60	55.40
Norway pout	*Trisopterus esmarki*	22.60	0.52	2.30	18.60
Poor cod	*Trisopterus minutus*	20.30	0.51	2.00	13.02
Hake	*Merluccius merluccius*	103.60	0.11	3.00	36.90
Angler	*Lophius piscatorius*	135.00	0.18	5.00	75.00
Grey gurnard	*Eutrigla gurnardus*	46.16	0.16	2.50	20.95
Bull rout	*Myoxocephalus scorpius*	34.00	0.24	2.00	15.00
Hooknose	*Agonus cataphractus*	17.40	0.42	2.00	9.22
Catfish	*Anarhichas lupus*	117.40	0.05	6.00	42.50
Dragonet	*Callionymus lyra*	22.20	0.47	1.50	13.29
Megrim	*Lepidorhombus whiffiagonis*	51.80	0.07	2.60	19.15
Turbot	*Scophthalmus maximus*	57.00	0.32	4.50	46.00
Witch	*Glyptocephalus cynoglossus*	45.50	0.16	3.00	20.00
Long rough dab	*Hippoglossoides platessoides*	24.60	0.34	2.60	15.14
Halibut	*Hippoglossus hippoglossus*	204.00	0.10	5.80	83.00
Common dab	*Limanda limanda*	26.70	0.26	2.25	13.08
Lemon sole	*Microstomus kitt*	37.10	0.42	4.00	27.00
Plaice	*Pleuronectes platessa*	54.40	0.11	2.50	26.60

5.3.1.3.5 Species trophic level

Information regarding the trophic level at which fish were feeding were available for 26 of the 56 species included in the SAGFS database (Jennings *et al.*, 2001, 2002). Variation in the trophic level at which fish were feeding was estimated by determining the stable nitrogen isotope ratios present in the white muscle tissue of fish sampled throughout the North Sea and Celtic Sea. Increase in the ^{15}N: ^{14}N ratio (henceforth referred to as the Nitrogen Ratio) reflects a higher trophic level diet (Minawaga and Wada, 1984). Relationships for Nitrogen Ratio at length were determined for a total of 31 species, of which 26 were encountered in the SAGFS database (Table 5.3.1.3.5.1). These 26 species accounted for over 98 % of the individuals sampled by the SAGFS in any of the spatial/temporal "treatments". Species abundance at length data were converted to the number of individuals with given Nitrogen Ratio values, and the mean value for each spatial/temporal "treatment" was computed.

Table 5.3.1.3.5.1. Parameters and test statistics for linear relationships between length (L mm $\log_{10}$ transformed) or weight (W g $\log_2$ transformed) and $\delta^{15}N$ $^0/_{00}$ or estimated trophic level (T.L.) of North Sea fishes. The form of the fitted relationships is $\delta^{15}N$ $^0/_{00}$ = a+ b($\log_{10}$ L). From Jennings *et al.* (in prep.).

Species	**Length ($\log_{10}$) vs. $\delta^{15}N$ $^0/_{00}$ and T.L.**						
	$\delta^{15}N$ $^0/_{00}$		**T.L.**				
	a	b	a	b	r^2	F	p
Wolfish	19.52	−2.53	6.08	-0.74	0.25	$1.4_{1,4}$	0.308
Scaldfish	13.47	0.96	4.31	0.28	0.32	$2.3_{1,5}$	0.190
Solenette	10.49	2.42	3.43	0.71	0.63	$20.1_{1,12}$	0.000
Dragonet	11.75	0.33	3.80	0.10	0.03	$0.1_{1,13}$	0.848
Four-bearded rockling	0.10	6.94	0.37	2.04	0.46	$12.8_{1,15}$	0.003
Grey gurnard	10.40	1.93	3.40	0.57	0.08	$1.5_{1,17}$	0.234
Cod	7.30	3.17	2.49	0.93	0.18	$4.9_{1,22}$	0.037
Witch	−7.38	7.92	−1.83	2.33	0.37	$5.3_{1,9}$	0.046
Long rough dab	10.82	0.95	3.53	0.28	0.02	$0.4_{1,19}$	0.550
Megrim	-8.82	7.90	−2.25	2.32	0.81	$37.7_{1,9}$	0.000
Dab	9.74	2.04	3.21	0.60	0.06	$2.1_{1,30}$	0.159
Monkfish	0.67	4.88	0.54	1.44	0.60	$14.9_{1,10}$	0.003
Haddock	8.63	2.24	2.88	0.66	0.20	$7.7_{1,30}$	0.010
Whiting	4.07	4.99	1.54	1.47	0.41	$5.5_{1,8}$	0.047
Hake	7.20	2.30	2.46	0.68	0.17	$1.2_{1,6}$	0.318
Lemon sole	6.04	3.10	2.12	0.91	0.14	$1.6_{1,10}$	0.236
Plaice	24.01	-3.89	7.43	−1.15	0.35	$10.8_{1,20}$	0.004
Saithe	−2.32	5.53	−0.34	1.63	0.71	$60.0_{1,23}$	0.000
Cuckoo ray	−3.37	6.12	−0.65	1.80	0.78	$57.6_{1,16}$	0.000
Starry ray	0.88	5.16	0.60	1.52	0.69	$28.4_{1,13}$	0.000
Lesser spotted dogfish	6.97	2.34	2.39	0.69	0.03	$0.14_{1,4}$	0.726
Norway haddock	−5.79	7.71	−1.36	2.27	0.87	$32.1_{1,5}$	0.002
Sole	2.20	5.48	0.99	1.61	0.32	$10.8_{1,23}$	0.003
Spurdog	−54.87	22.97	−15.80	6.76	0.90	$34.0_{1,4}$	0.004
Norway pout	1.51	5.25	0.79	1.54	0.53	$18.0_{1,16}$	0.001
Poor cod	21.28	−2.61	6.60	−0.77	0.06	$1.0_{1,15}$	0.342

5.3.1.4 Analysis and results

In this section we examine each of the principal theoretical statements posed at the beginning of this report. The section is therefore divided into three sub-sections, each devoted to one of the theoretical statements. In each sub-section we pose four explicit hypotheses, two concerning the contemporary condition of the groundfish assemblage, and two concerning long-term changes in the groundfish assemblage, and we then examine the data to determine the extent to which each hypothesis is supported or refuted.

5.3.1.4.1 Effects of fishing on species richness and species diversity

Our hypotheses regarding species richness and species diversity of the groundfish assemblage in contemporary times are stated as:

- Groundfish assemblage species richness and species diversity should be highest, in rectangles where current levels of otter trawling are lowest, and least in rectangles where current levels of otter trawling are highest.
- Groundfish assemblage species richness and species diversity should be highest, in rectangles where the rate of increase in otter trawling over recent decades has been slowest, or where otter trawling levels have declined, and least in rectangles where otter trawling levels have increased fastest over recent decades.

Mean (± 1 S.E. of the mean) species richness, and Hill's (1973) N1 and N2 species diversity indices, were determined for the three groups of rectangles varying in their current exploitation levels, and for the three groups of rectangles varying in their rates of change in otter trawling levels in recent decades (Figure 5.3.1.4.1.1). Differences, tested using one-way ANOVA, were found to be significant at the 1 % level for all three community structure metrics in each of the two investigations.

In both investigations variation in species richness across the three groups of rectangles appeared to refute the stated hypotheses. Species richness was lowest where current levels of otter trawling are low, and where otter trawling has been declining or increasing slowly over recent decades (the difference between these two treatments was not significant). Species richness was highest in rectangles of medium and high current levels of otter trawling (the difference between these two treatments was not significant) and where otter trawling has increased at the fastest rate over recent decades. Either fishing has caused an increase in species richness, or fishing has increased most in areas where species richness is highest. Variation in species diversity supported the hypothesis and both indices behaved in an identical manner. The difference in species diversity between areas of medium and high current otter trawling intensity was marginal, and not statistically significant. These indices may be sensitive to changes in the groundfish assemblage brought about by low levels of fishing perturbation, but were unable to differentiate between moderately and heavily disturbed communities. Both indices, however, appeared to be sensitive to the full range of annual rates of change in fishing activity, differentiating between all three groups of rectangles.

In order to discount the possible alternative interpretation of these results, i.e., that fishing has been attracted to areas of low species richness and diversity, our hypotheses regarding temporal trends in species richness and species diversity of the groundfish assemblage over the period 1925 to 1996 are therefore stated as:

- Slopes of the temporal trend of species richness and species diversity of the groundfish assemblage should be most steeply negative in rectangles where current levels of otter trawling are highest, and less steep, or zero, in rectangles where current levels of otter trawling are lowest.
- Slopes of the temporal trend of species richness and species diversity of the groundfish assemblage should be most steeply negative in rectangles where, over recent decades, annual rates of increase in otter trawling levels have been greatest. In rectangles where otter trawling levels have declined, the slopes of the long-term trend in species richness and species diversity should be zero, or even positive.

This analysis used the full time series of available groundfish survey data, from 1925 to 1996. Long-term variation in the species richness and species diversity of the groundfish assemblage in rectangles varying in their current levels of fishing exploitation, and in rectangles differing in their recent temporal trends in fishing effort, was examined. The rectangles were grouped into the same three treatment levels of current international otter trawl effort during the early 1990s, and the same three treatments of trends in Scottish otter trawl effort over the period 1970 to 1994. Groundfish community species richness and diversity were determined for each year-group for all the rectangles assigned to each "current effort level" treatment and each "recent decades effort trend" treatment. These were then regressed over time and the regression coefficients (± 1 S.E. of the coefficient) were plotted for each treatment and life-history characteristic (Figure 5.3.1.4.1.2).

Figure 5.3.1.4.1.1. Variation in the mean (± 1S.E.) groundfish assemblage species richness, and Hill's (1973) N1 and N2 species diversity indices, calculated for three groups of rectangles varying in the level of otter trawl effort to which they are currently subjected (A), and for three groups of rectangles varying in their annual rates of change in otter trawl effort over recent decades (B).

A **B**

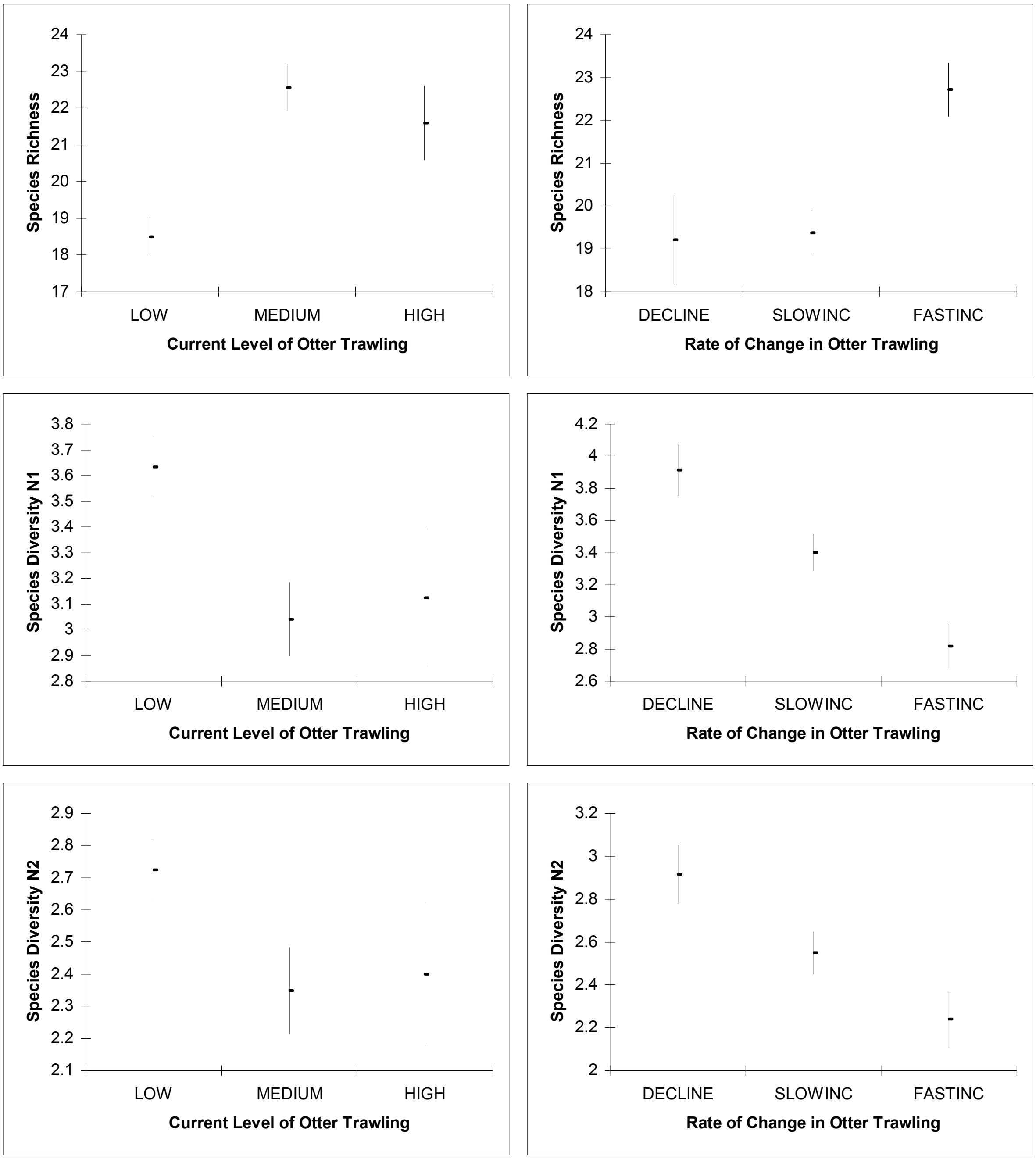
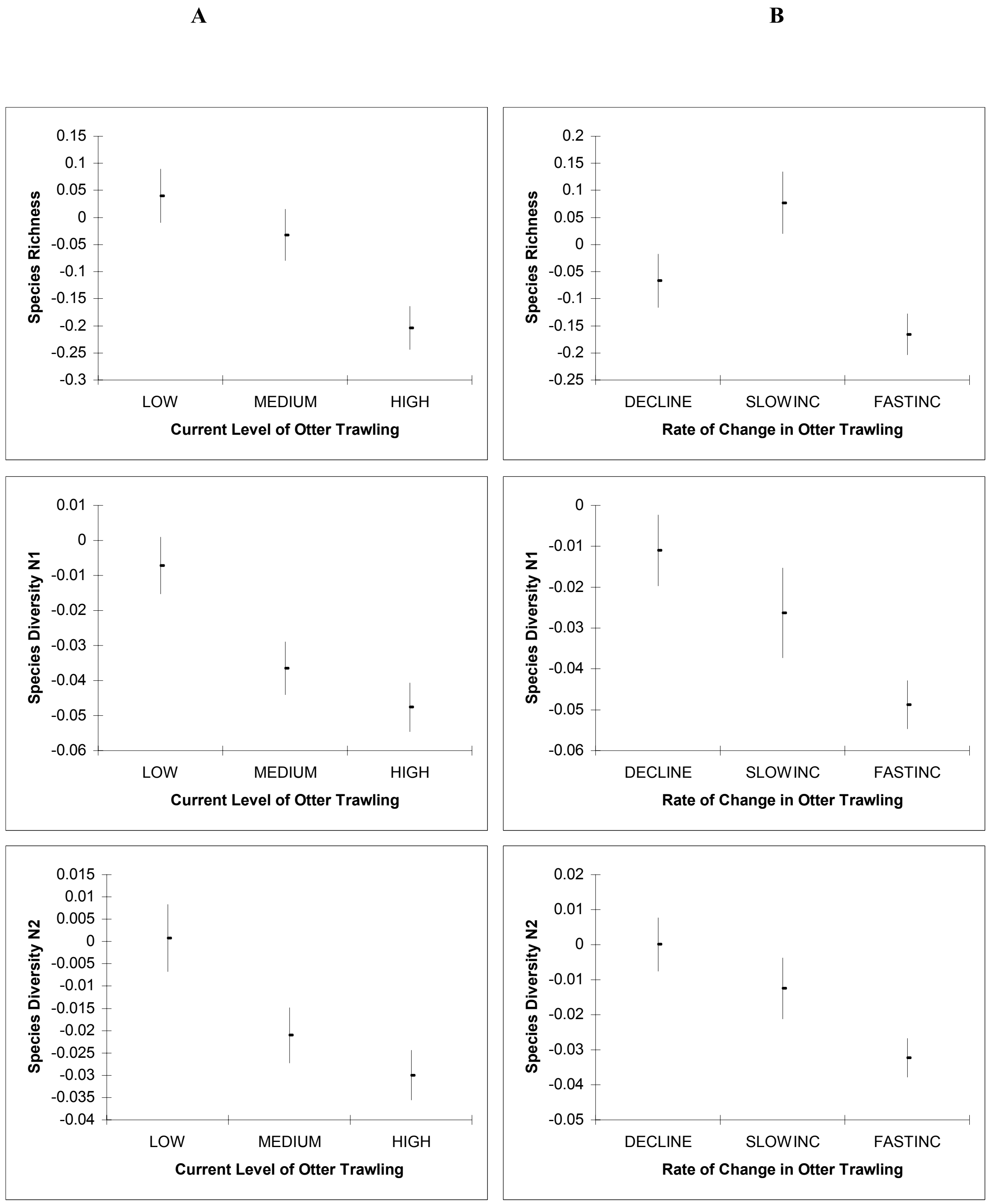


Figure 5.3.1.4.1.2. Variation in the regression coefficients (± 1S.E. of the coefficient) for the slopes of groundfish assemblage species richness, and Hill's (1973) N1 and N2 species diversity indices, determined over the period 1925 to 1996 for three groups of rectangles varying in the level of otter trawl effort to which they are currently subjected (A), and for three groups of rectangles varying in their annual rates of change in otter trawl effort over recent decades (B).



With regard to current exploitation levels, all three metrics supported the hypothesis. Long-term trends in rectangles where fishing levels were low showed little or no trend, but were increasingly negative with increasing levels of disturbance. Their response to recent trends in perturbation also tended to support the hypothesis. Both species diversity indices showed little or no trend where effort had declined, but in rectangles where effort had increased slowly negative trends were observed, and these trends became more negative in rectangles where effort had increased at a faster rate. Species richness, however, did not behave entirely as expected. Steep negative trends were observed in areas where effort had increased fastest, and little or no trend was detected where effort had declined. However, in rectangles where effort had increased slowly, slightly positive trends in species richness were observed.

5.3.1.4.2 Effects of fishing on groundfish assemblage life history characteristics

Our hypotheses regarding the life-history characteristics exhibited by the groundfish assemblage in contemporary times are stated as:

- Averaged over the whole groundfish assemblage, contemporary Growth Rates should be highest, and $\text{Length}_{\text{Infinity}}$, $\text{Length}_{\text{Maturity}}$ and $\text{Age}_{\text{Maturity}}$ should be lowest in rectangles where current levels of otter trawling are highest.
- Averaged over the whole groundfish assemblage, contemporary Growth Rates should be highest, and $\text{Length}_{\text{Infinity}}$, $\text{Length}_{\text{Maturity}}$ and $\text{Age}_{\text{Maturity}}$ should be lowest in rectangles where, over recent decades, annual rates of increase in otter trawling levels have been highest.

Mean (± 1 S.E. of the mean) contemporary life-history characteristic values were determined for the three groups of rectangles varying in their current exploitation levels, and for the three groups of rectangles varying in their rates of change in otter trawling levels in recent decades (Figure 5.3.1.4.2.1). Differences, tested using one-way ANOVA, were found to be significant at the 1% level for all four characteristics in each of the two investigations.

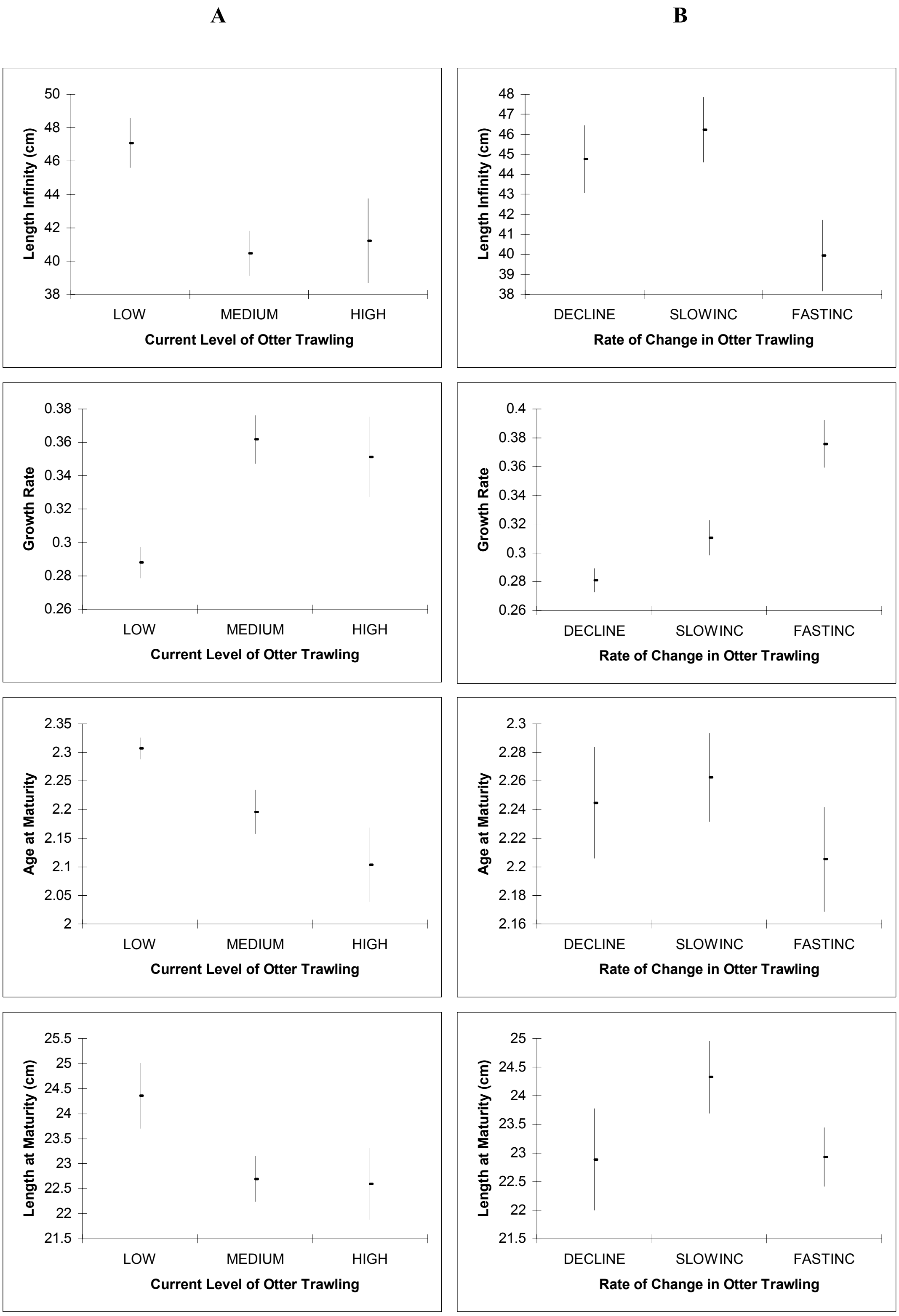
With regard to current exploitation levels, for each life-history characteristic the data supported the hypothesis. $\text{Length}_{\text{Infinity}}$, Growth Rates, and $\text{Length}_{\text{Maturity}}$ appeared to be the most sensitive characteristics, differentiating most between low and medium fishing intensity groups of rectangles. Beyond a certain (threshold?) level of perturbation, variation in these characteristics tended to level out. This raises the possibility that, as metrics, these three characteristics may be relatively insensitive in already disturbed areas. On the other hand, they continued to decrease strongly as otter trawl effort increased from low, through medium, to high levels.

The situation was less clear-cut with respect to the effect of longer-term change in otter trawling levels on contemporary mean groundfish assemblage life-history characteristics. Only for Growth Rate did the data appear to fully support the hypothesis. $\text{Length}_{\text{Infinity}}$, $\text{Length}_{\text{Maturity}}$ and $\text{Age}_{\text{Maturity}}$ all differentiated between rectangle where otter trawling levels had increased slowly over recent decades and rectangles where exploitation had increased quickly in a way that was anticipated by the hypothesis. However, for all three life-history characteristics, the mean assemblage values in rectangles where fishing was actually declining were lower than in rectangles where exploitation was increasing slowly. This result was unanticipated by the hypothesis.

In order to discount the possible alternative interpretation of these results, that fishing has been attracted to areas with communities consisting of fish with small $\text{Length}_{\text{Infinity}}$, and $\text{Length}_{\text{Maturity}}$, young $\text{Age}_{\text{Maturity}}$, and fast Growth Rates, our hypotheses regarding the temporal trends in the life-history characteristics exhibited by the groundfish assemblage over the period 1925 to 1996 are stated as:

- Averaged over the whole groundfish assemblage, temporal trends in Growth Rates should have the steepest positive slopes, and temporal trends in $\text{Length}_{\text{Infinity}}$, $\text{Length}_{\text{Maturity}}$ and $\text{Age}_{\text{Maturity}}$ should have the steepest negative slopes, in rectangles where current levels of otter trawling are highest.
- Averaged over the whole groundfish assemblage, trends in Growth Rates should be more positive, and temporal trends in $\text{Length}_{\text{Infinity}}$, $\text{Length}_{\text{Maturity}}$ and $\text{Age}_{\text{Maturity}}$ should be more negative, in rectangles where, over recent decades, annual rates of increase in otter trawling levels have been greatest. In rectangles where otter trawling levels have actually declined, long-term trends in life-history characteristics should be zero, or in the opposite direction.

Figure 5.3.1.4.2.1. Variation in the mean (± 1S.E.) $Length_{Infinity}$, Growth Rate, $Age_{Maturity}$ and $Length_{Maturity}$, determined for 28 species making up >98 % of the total number of individual groundfish sampled in each rectangle, calculated for three groups of rectangles varying in the level of otter trawl effort to which they are currently subjected (A), and for three groups of rectangles varying in their annual rates of change in otter trawl effort over recent decades (B).



This analysis used the full time series of available groundfish survey data, from 1925 to 1996. Long-term variation in the mean life-history characteristics of the groundfish assemblage in rectangles varying in their current levels of fishing exploitation, and in rectangles differing in their recent temporal trends in fishing effort, was examined. The rectangles were grouped into the same three treatment levels of current international otter trawl effort during the early 1990s, and the same three treatments of trends in Scottish otter trawl effort over the period 1970 to 1994. Abundance-weighted mean life-history character values for the groundfish community were determined for each year-group for all the rectangles assigned to each "current effort level" treatment and each "recent decades effort trend" treatment. These were then regressed over time and the regression coefficients (± 1 S.E. of the coefficient) were plotted for each treatment and life-history characteristic (Figure 5.3.1.4.2.2).

With regard to current exploitation levels, all four parameters showed very little change in rectangles where levels of otter trawl impact were low. Indeed, none of the long-term regression analyses were significant. However, in rectangles with medium and high levels of international otter trawl effort during the early 1990s, all the long-term trends were significant, and in the direction predicted by our hypotheses. Of interest once more was the fact that $\mathrm{Length}_{\mathrm{Infinity}}$, Growth Rate and $\mathrm{Length}_{\mathrm{Maturity}}$ all failed to differentiate between medium and high levels of fishing effort. This again suggests that these parameters may be able to distinguish between fished and unfished areas, but once an area is impacted, it may be relatively insensitive to further perturbation. $\mathrm{Age}_{\mathrm{Maturity}}$, however, showed increasingly steep long-term declines as otter trawl effort increased from medium to high levels of otter trawl activity. This analysis therefore appears to confirm the possibility that this index may hold the greatest promise as a metric able to provide managers with an ongoing indication of the continuing effect of their actions on the life-history composition of the groundfish community.

Once again, the situation was less clear-cut with respect to the effect of change in otter trawling levels over recent decades on long-term trends in mean groundfish assemblage life-history characteristics. The greatest long-term rates of decline in both $\mathrm{Age}_{\mathrm{Maturity}}$ and $\mathrm{Length}_{\mathrm{Maturity}}$ occurred in rectangles where otter trawl activity has actually declined over the period 1970 to 1994. This clearly contravenes the hypothesis. $\mathrm{Length}_{\mathrm{Maturity}}$ does at least show steeper long-term declines in rectangles where otter trawling has increased most rapidly over the period 1970 to 1994, compared with rectangles where trawling has increased slowly. $\mathrm{Age}_{\mathrm{Maturity}}$ fails even to do this. Variation in $\mathrm{Length}_{\mathrm{Infinity}}$ and Growth Rate, however, both support the hypothesis. The long-term decline in $\mathrm{Length}_{\mathrm{Infinity}}$ and long-term increase in Growth Rate are both steepest in the rectangles where otter trawling has increased the quickest over the period 1970 to 1994, compared with rectangles where trawling has increased slowly. The trends in $\mathrm{Length}_{\mathrm{Infinity}}$ and Growth Rate in rectangles where fishing had declined did not differ significantly from trends in rectangles where fishing had increased slowly.

5.3.1.4.3 Effects of fishing on groundfish assemblage trophic structure

Our hypotheses regarding the trophic structure of the groundfish assemblage in contemporary times are stated as:

- The trophic level of fish belonging to the groundfish assemblage should be higher, on average, in rectangles where current levels of otter trawling are lowest, and lower in rectangles where current levels of otter trawling are highest.
- The trophic level of fish belonging to the groundfish assemblage should be higher, on average, in rectangles where the rate of increase in otter trawling over recent decades has been slowest, and lower in rectangles where otter trawling levels have increased fastest over recent decades.

Mean (± 1 S.E. of the mean) contemporary nitrogen ratio values were determined for the three groups of rectangles varying in their current exploitation levels, and for the three groups of rectangles varying in their rates of change in otter trawling levels in recent decades (Figure 5.3.1.4.3.1). Differences, tested using one-way ANOVA, were found to be significant at the 1 % level in each of the two investigations. No significant differences in the mean nitrogen ratio in the demersal fish community occupying rectangles differing in their current levels of fishing disturbance were detected. However, the trophic level at which demersal fish were feeding was significantly higher in rectangles where effort was declining compared with that observed for rectangles where effort had increased slowly, which in turn was higher than the trophic level observed in rectangles where effort had increased at the fastest rate.

Figure 5.3.1.4.2.2. Variation in the regression coefficients (± 1S.E. of the coefficient) for the slopes of $Length_{Infinity}$, Growth Rate, $Age_{Maturity}$ and $Length_{Maturity}$, determined over the period 1925 to 1996 for 28 species making up >98 % of the total number of individual groundfish sampled in each rectangle, for three groups of rectangles varying in the level of otter trawl effort to which they are currently subjected (A), and for three groups of rectangles varying in their annual rates of change in otter trawl effort over recent decades (B).

A **B**

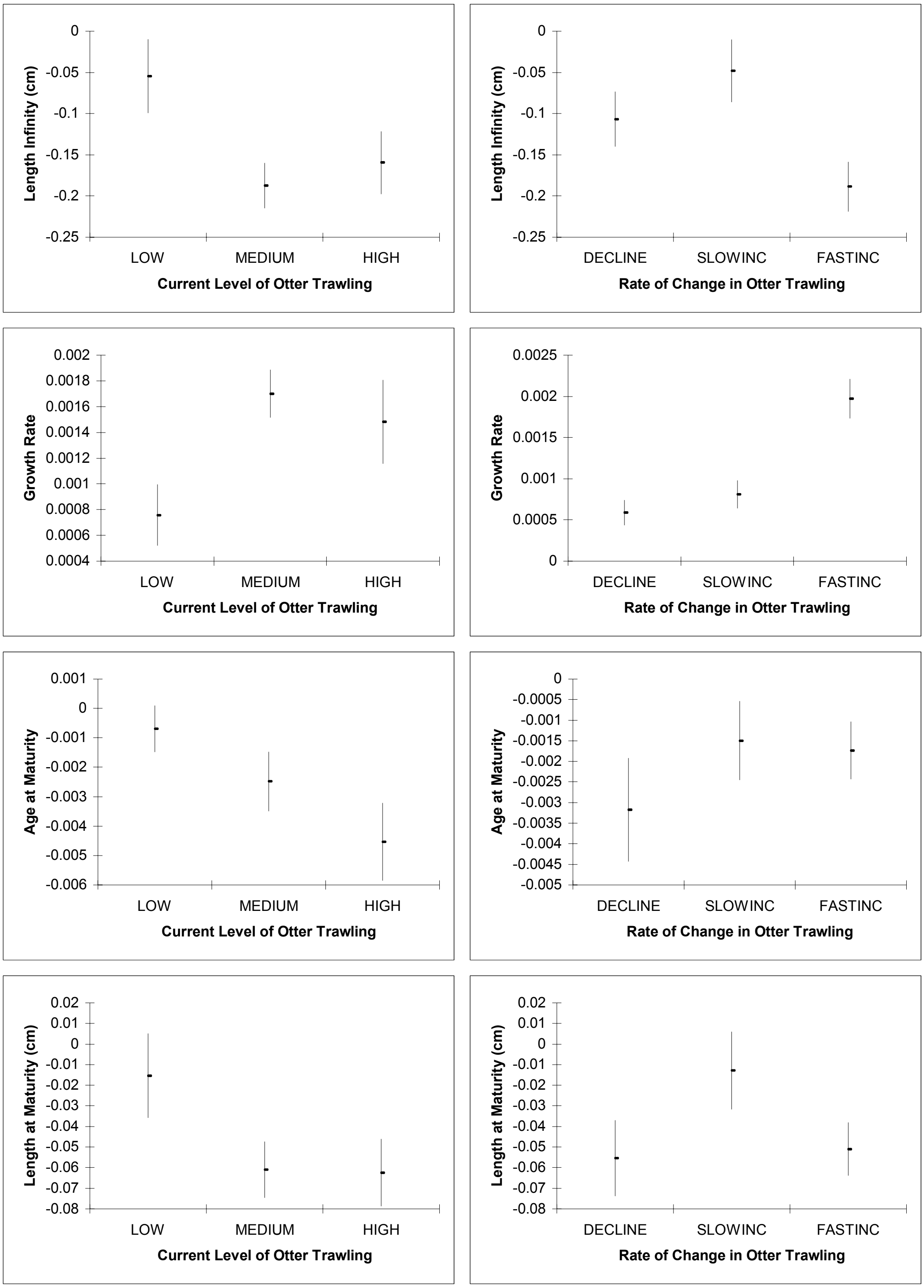


Figure 5.3.1.4.3.1. Variation in the mean (± 1S.E.) Nitrogen Ratio for 26 species making up >98 % of the total number of individual groundfish sampled in each rectangle, calculated for three groups of rectangles varying in the level of otter trawl effort to which they are currently subjected (A), and for three groups of rectangles varying in their annual rates of change in otter trawl effort over recent decades (B).

A B

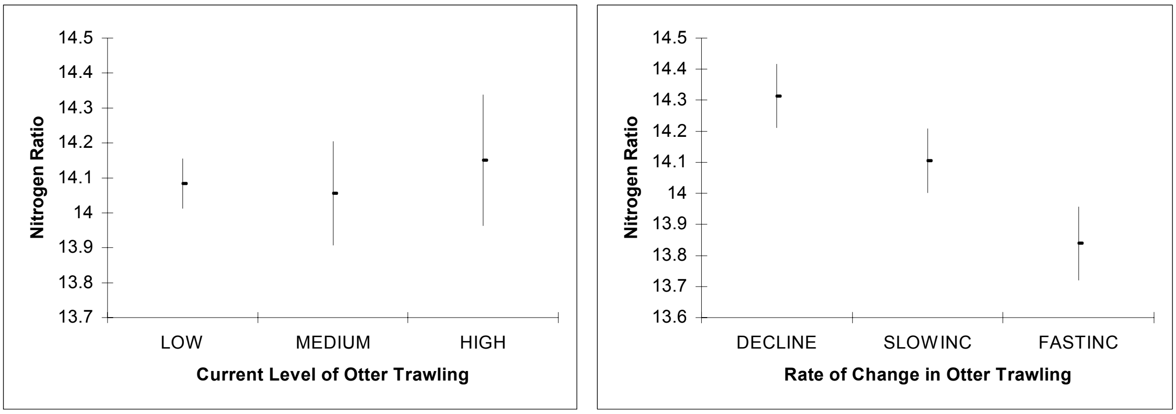
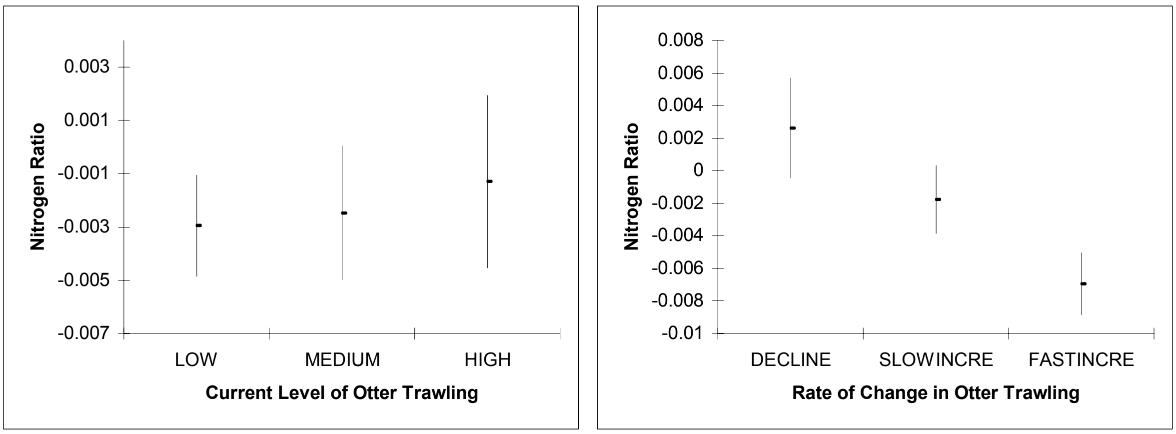


Figure 5.3.1.4.3.2. Variation in the regression coefficients (± 1S.E. of the coefficient) for the slopes of Nitrogen Ratio determined over the period 1925 to 1996 for 26 species making up >98 % of the total number of individual groundfish sampled in each rectangle, for three groups of rectangles varying in the level of otter trawl effort to which they are currently subjected (A), and for three groups of rectangles varying in their annual rates of change in otter trawl effort over recent decades (B).

A B



In order to discount the possible alternative interpretation of these results, that fishing has been attracted to areas with fish that feed at low trophic levels, our hypotheses regarding trends in the trophic structure of the groundfish assemblage over the period 1925 to 1996 are stated as:

- Slopes of the temporal trend in the mean trophic level of fish belonging to the groundfish assemblage should be most steeply negative in rectangles where current levels of otter trawling are highest, and less steep, or zero, in rectangles where current levels of otter trawling are lowest;
- Slopes of the temporal trend in the mean trophic level of fish belonging to the groundfish assemblage should be most steeply negative in rectangles where, over recent decades, annual rates of increase in otter trawling levels have been greatest. In rectangles where otter trawling levels have declined, slopes of the long-term trend in mean trophic level should be zero, or even positive.

This analysis used the full time series of available groundfish survey data, from 1925 to 1996. Long-term variation in the mean nitrogen ratio of the groundfish assemblage in rectangles varying in their current levels of fishing exploitation, and in rectangles differing in their recent temporal trends in fishing effort, was examined. The rectangles were grouped into the same three treatment levels of current international otter trawl effort during the early 1990s, and the same three treatments of trends in Scottish otter trawl effort over the period 1970 to 1994. Abundance-weighted mean nitrogen ratio values for the groundfish community were determined for each year-group for all the rectangles assigned to each "current effort level" treatment and each "recent decades effort trend" treatment. These were then regressed over time and the regression coefficients (± 1 S.E. of the coefficient) were plotted for each treatment (Figure 5.3.1.4.3.2).

Little or no trend in mean groundfish assemblage nitrogen ratios was observed in rectangles differing in their current exploitation levels. Once again, however, negative trends in nitrogen ratio, implying a decline in trophic level, were observed in the groundfish assemblage occupying rectangles where effort had increased at the fastest rate. In rectangles where effort had declined, or increased only slowly, little or no long-term trend in nitrogen ratio could be detected.

5.3.1.5 Discussion

Table 5.3.1.5.1 summarises the above results. From this table it can be seen that these hypotheses are generally supported by the data, strengthening the case that fishing is responsible for the changes observed in the groundfish assemblage. Explanations for why, in some cases, the hypotheses have not been supported by the data are given below. The table also indicates the types and levels of disturbance for which each metric could potentially act as an indicator.

Table 5.3.1.5.1. Summary of community metric behaviour.

Community characteristic or metric	Behaved as predicted to		Potential indicator of / between		
	Current fishing effort	**Rates of change in fishing effort**	**Fished / unfished areas**	**Different levels of fishing effort**	**Rates of change in fishing effort**
Species Richness	No	No	-	-	-
Species Diversity	Yes	Yes	Yes	No	Yes
$Length_{Infinity}$	Yes	Yes	Yes	No	Yes
Growth Rate	Yes	Yes	Yes	No	Yes
$Age_{Maturity}$	Yes	Yes	Yes	Yes	Yes
$Length_{Maturity}$	Yes	Yes	Yes	No	Yes
Nitrogen Ratio	No	No	-	-	-

Species richness did not respond to either treatment. It is greatest in rectangles with medium and high current levels of fishing effort. This, combined with the observation that there is long-term species richness decline in rectangles with high fishing effort and in rectangles where effort had increased fastest, may mean that fishermen are concentrating their activities in rectangles where groundfish species richness is higher.

Species diversity, $Length_{Infinity}$, Growth Rate, $Age_{maturity}$ and $Length_{maturity}$ all responded to the treatments as predicted by the hypotheses. The two diversity indices appear capable of distinguishing between relatively undisturbed and disturbed communities, but appear insensitive to variations in disturbance above a certain disturbance threshold. They would therefore not be useful as potential indicators of different levels of fishing effort. Although mean $Length_{Infinity}$, Growth Rate and $Length_{maturity}$ were lower in areas of medium and high current fishing effort, these metrics were unable to differentiate between these two treatments. Thus, while these metrics are also capable of distinguishing between relatively undisturbed and disturbed communities, they are insensitive to variation in disturbance above a certain disturbance threshold. They would not therefore be useful as potential indicators of different levels of fishing effort either. Mean $Age_{Maturity}$ differentiated between all three current fishing disturbance treatments, and so could potentially provide a good indicator of changes in the level of perturbation, as it is sensitive across the entire range of current effort treatments considered here.

No difference in the mean assemblage Nitrogen ratios was observed for any of the three current effort treatments, thus this metric did not respond as predicted by this particular hypothesis. However, Nitrogen ratio data suggested that fish in rectangles where fishing effort had declined over recent decades were feeding at a higher trophic level than fish in rectangles where effort had increased slowly. These were, in turn, feeding at a higher trophic level than fish in rectangles where effort had increased at the fastest rates. These results therefore support the hypothesis. Data for the long-term trends showed identical patterns. At first sight these results appear contradictory, however, they can be interpreted as suggesting that fishermen are directing their efforts to areas where fish are feeding at a higher trophic level. This has disturbed the community such that rates of decline in Nitrogen ratio have been fastest in areas where fishing disturbance has been the greatest. As a result the trophic level difference between the groundfish assemblages can no longer be detected.

The analytical design makes a major assumption, that the communities present in the rectangles assigned to the different treatment groups all had the same start point with respect to each of the community characteristics. In the cases of species richness and community trophic structure, where some deviation from the hypotheses' predictions were observed, these discrepancies have been explained on the basis that this assumption was violated; that fishing activities have indeed been attracted in the first place to areas of high species richness and a high trophic level assemblage. There is some theoretical basis for such an assertion. It seems entirely plausible that fishing might be more profitable in high productivity areas, which might be expected to support higher densities of fish, and there is a considerable body of theoretical literature linking species richness to productivity (see references in Huston, 1994; Davidson, 1977; Rosenzweig, 1992, 1995; Rosenzweig and Abramsky, 1993). Higher trophic level has frequently been linked to large body size (see references in Jennings *et al.*, 2001, 2002), and fishing activities have always been directed towards the larger fish in the assemblage. Under such circumstances, actually confirming a fishing impact on the assemblage will be exceptionally difficult because information regarding the precise conditions of the community will rarely be available. The fact that, for these two parameters at least, the communities may well have been different prior to fishing means that we should consider implications of this with regard to our interpretation of the results for the remaining community characteristics: those where the data support our hypotheses, suggesting that fishing was the cause of the observed changes in the community. There was insufficient time available during the WGECO meeting for such considerations, but this issue will be addressed in due course.

There are close interrelationships between life history parameters, thus any effects of fishing on mean $Length_{maturity}$, $Age_{maturity}$, $Length_{infinity}$ and Growth Rate would tend to be correlated. In an analysis of the type we present, where fixed life history parameters are assigned to all individuals in the community, the change in the value of the life history parameter simply reflects a change in the species composition of the community. As the community is increasingly dominated by species with fast life histories that are less vulnerable to fishing, so mean $Length_{maturity}$, $Age_{maturity}$ and $Length_{infinity}$ would be expected to fall and Growth Rate to rise. These are indeed the responses we observe, but it is clear that the mean values of some life histories appear to show a stronger response to fishing than others. We assume that this is a function of the extent to which the life history parameters can discriminate species with different responses to fishing in the community. Thus, $Length_{maturity}$, $Length_{infinity}$ and Growth Rate can almost be regarded as continuous variables which distinguish all species, while $Age_{maturity}$ tends to be a categorical variable because $Age_{maturity}$ is often reported as an integer value.

This analysis, as with other analyses of the impacts of fishing on marine communities, does not allow us to separate the first and second order effects of fishing (see Text Box 1). For this reason, the responses of species richness, diversity, mean life history parameters and trophic level to fishing may appear to be correlated, when the responses are actually independent. For example, the mean maximum size ($Length_{infinity}$) of the entire community may fall in response to the direct effects of size-selective fishing mortality because species with larger body size will i) suffer higher mortality; and ii) have less capacity to tolerate this mortality (first order effect). However, the diversity of the community may fall because high levels of fishing effort reduce the heterogeneity of seabed habitats (second order effect). At present, we have not developed methodologies that adequately disentangle the first and second order effects of fishing.

5.3.2 Comparative impacts of bottom fishing on trophic structure and size composition in the North Sea

5.3.2.1 Introduction

In this analysis, we examine long-term changes in the trophic structure of the North Sea fish community using four time series of species-size-abundance trawl survey data. Several surveys were used to generate the time series presented. Three ongoing surveys are conducted on a routine basis in the North Sea: the International Bottom Trawl Survey (IBTS) which has covered the whole of the North Sea, the BTS, and the SNS. The Scottish groundfish survey (1925–1996) was terminated in 1996 and covers the central and northern North Sea. The surveys differ in the type of gear, spatial coverage and temporal coverage and can therefore be considered complementary.

For comparison between surveys, time series were generated for a number of metrics describing various aspects of the fish community: trophic level, mean weight, slope biomass size spectra, mean maximum length, and biodiversity indices Hill's N0, N1 and N2. Since all surveys are designed to sample the demersal fish assemblage, pelagic species were excluded from the analyses. For more detailed analyses on the effects of fishing, two suites of species were distinguished: commercial species and non-target species. Time series were generated for each of these suites separately. In order to compare the trends between surveys for each suite of fish species, linear regression was used to determine the trend for each of the metrics.

Rather than assigning fixed trophic levels to species that can vary in size by orders of magnitude during the course of their life history, we determined relationships between trophic level and body size for each species and applied these to the species-size-abundance data to estimate the mean trophic level of the community. We also examined relationships between mean trophic level, mean body size, mean maximum body size and the slopes and intercepts of biomass size-spectra. If such relationships can be established, mean body size or the slopes and intercepts of biomass size-spectra may provide a convenient and easily measured metric to assess the relative impacts of fishing on trophic structure (Rice, 2000; Piet, 2001). In the EcoQO framework the Hill numbers N_0 (species richness), N_1 (exponential of Shannon-Wiener's diversity index, effectively the number of abundant species) and N_2 (reciprocal of Simpson's diversity index, effectively the number of very abundant species) were used to describe the diversity of the fish community. These metrics are further explored in this section.

5.3.2.2 Methods to estimate metrics

Estimating trophic level from $\delta^{15}N$

An alternative method for assessing trophic level is nitrogen stable isotope analysis, because the abundance of $\delta^{15}N$ in the tissues of predators is typically 3.4 ‰ greater than that in the tissues of their prey (Minawaga and Wada, 1984; Owens, 1987). Therefore, if the $\delta^{15}N$ of organisms at the base of the food chain is known, and these organisms can be assigned to a trophic level, the trophic level of organisms higher in the food chain can be predicted (Owens, 1987).

To assign trophic level to species on the basis of body size, we used relationships for North Sea fishes sampled in 2001 (Table 5.3.1.3.5.1). To allow for comparisons between our study and other assessments of the effects of fishing on trophic structure, we converted some of our $\delta^{15}N$ values to trophic level. This required an assessment of the trophic level of animals close to the base of the food chain. We used bivalve molluscs that are filter and suspension feeders. We used linear regression to describe intraspecific relationships between length or weight and $\delta^{15}N$. Lengths (in mm) were $\log_{10}$ transformed. $\delta^{15}N$ values were converted to trophic level based on the assumption that there was a fractionation of +3.4 ‰ per trophic level (Minagawa and Wada, 1984) and that the base material (bivalve molluscs) had a trophic level of 2.5

Trophic structure

We applied our estimates of $\delta^{15}N$ and trophic level to species-size-abundance data from trawl survey catches. For the IBTS, we identified all ICES rectangles where at least one Grande Ouverture Verticale (GOV) trawl survey haul had been made every year from 1982–2000. There were 110 such rectangles. We calculated the mean annual catch (numbers) per 30-minute tow for these rectangles by species and length class. For the SGFS, we calculated mean catch rates by species and size class for 19 groups of years that covered the period 1925 to 1996 in the central and northern North Sea. Year groups ranged from 1 to 4 consecutive years and were selected to ensure that the spatial coverage of the combined hauls was similar in each group. Greenstreet *et al.* (1999a) indicated that it was essential to group years in this way because significant spatial variation in catch rates could obscure temporal trends in community structure. Overall fishing effort in the study region has increased since 1960 and was also assumed to have increased steadily before 1960 due to continued improvements in the seaworthiness of vessels and the development of offshore fisheries (Greenstreet *et al.*, 1999b).

The $\delta^{15}N$ of all individuals of the 31 study species in the catches was estimated from their lengths using the relationship between $\delta^{15}N$ and log_{10} length. We calculated the weighted mean $\delta^{15}N$ for catches in each year group. We excluded all fish <4 g from the analysis as these fish are poorly sampled by the survey trawls.

For the IBTS data we calculated the weighted mean $\delta^{15}N$ for i) all demersal (bottom dwelling) and pelagic species sampled by the trawl; and ii) for demersal species only. We classified herring *Clupea harengus*, mackerel *Scomber scombrus*, horse mackerel *Trachurus trachurus*, sandeel *Ammodytes marinus,* and sprat *Sprattus sprattus* as pelagic species and all others as demersal. The Aberdeen otter trawl used for the SGFS does not sample pelagic species effectively and so we only included demersal species in the SGFS data analysis.

Biomass size spectra

Biomass size spectra were calculated for the fish communities sampled in each year (IBTS) or year group (SGFS). Fish >4 g were assigned to log_2 body mass classes, and cumulative biomass by log_2 body mass was calculated. Biomass size spectra were normalised by dividing the biomass in a given body mass class interval by the width of that class interval. The relationship between body mass (as log_2 classes) and total normalised biomass was described using least squares linear regression. Mean body size and mean maximum body size were calculated as mean log_2 body mass and mean log_2 of maximum body mass respectively (Table 5.3.2.2.1).

Table 5.3.2.2.1. The scientific and common names, maximum recorded lengths and weights of North Sea fishes. From Jennings *et al.* (2001) except for spurdog *Squalus acanthias* (new data).

Scientific name	Common Name	Maximum length (mm)	Maximum weight (g)
Ammodytes marinus	Raitt's sandeel	240	48
Anarhichas lupus	wolfish	1,000	10,392
Arnoglossus laterna	scaldfish	160	38
Buglossinium luteum	solenette	130	22
Callionymus lyra	dragonet	300	148
Clupea harengus	herring	340	326
Enchelyopus cimbrius	four-bearded rockling	330	182
Eutrigla gurnardus	grey gurnard	460	886
Gadus morhua	cod	1,230	17,650
Glyptocephalus cynoglossus	witch	470	696
Hippoglossoides platessoides	long rough dab	250	133
Lepidorhombus whiffiagonis	megrim	610	2,059
Limanda limanda	dab	330	400
Lophius piscatorius	monkfish	1,060	18,045
Melanogrammus aeglefinus	haddock	720	3,515
Merlangius merlangus	whiting	540	1,360
Merluccius merluccius	hake	1,100	10,950
Microstomus kitt	lemon sole	457	1,181
Pleuronectes platessa	plaice	580	2,157
Pollachius virens	saithe	1,550	23,609
Raja naevus	cuckoo ray	920	4,220
Raja radiata	starry ray	660	2,450
Scomber scombrus	mackerel	399	555
Scyliorhinus canicula	lesser spotted dogfish	880	2,763
Sebastes vivparous	Norway haddock	360	876
Solea solea	sole	460	950
Sprattus sprattus	sprat	170	38
Squalus acanthias	spurdog	1,100	8,500
Trachurus trachurus	horse mackerel	500	1,344
Trisopterus esmarki	Norway pout	230	122
Trisopterus minutus	poor cod	420	1,095

For 18 of the 31 fish species, relationships ($p < 0.05$) between $\delta^{15}N$ or trophic level and length were significant (Table 5.3.1.3.5.1). All the significant relationships were positive, except those for herring *Clupea harengus* and plaice *Pleuronectes platessa*, which appeared to feed at lower trophic levels as they increased in size.

5.3.2.2.1 Trends in trophic level

The mean $\delta^{15}N$ of the whole fish community sampled during the IBTS decreased significantly from 1982–2000 (Figure 5.3.2.2.1.1; Sens non-parametric test of slope, $p<0.05$). Changes in the mean $\delta^{15}N$ of the demersal fish community were not significant in the same period (Figure. 5.3.2.2.1.1, Sens test $p>0.05$). The mean $\delta^{15}N$ of the demersal fish community sampled on the SGFS remained relatively stable from 1925 to 1996 (Figure 5.3.2.2.1.2).

Figure 5.3.2.2.1.1 Long-term trends in the mean $\delta^{15}N$ and equivalent trophic level of the North Sea fish community, as sampled by the IBTS. Pelagic and demersal species = filled circles. Demersal species = open circles.

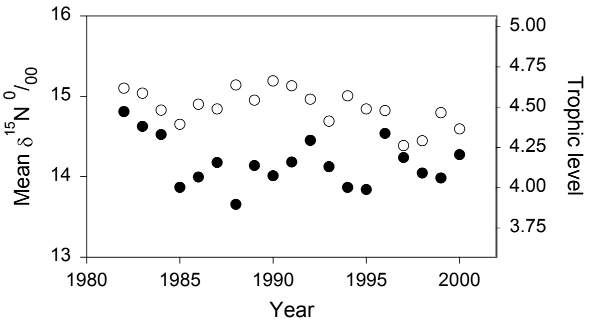


Figure 5.3.2.2.1.2. Long-term trends in the mean $\delta^{15}N$ and equivalent trophic level of the North Sea demersal fish community, as sampled by the SGFS.

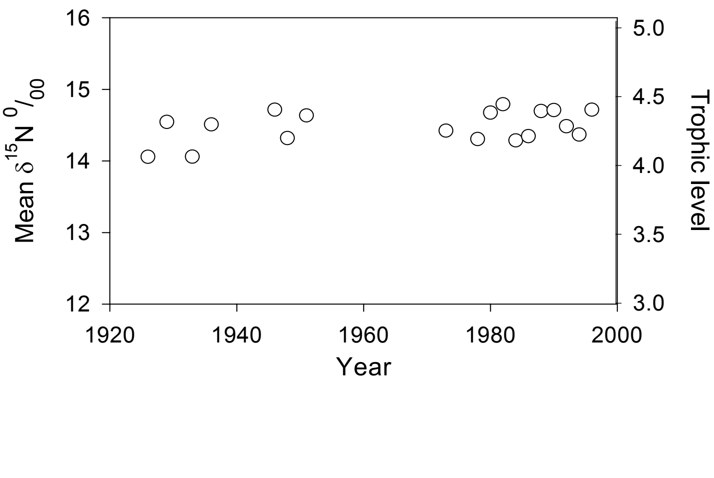
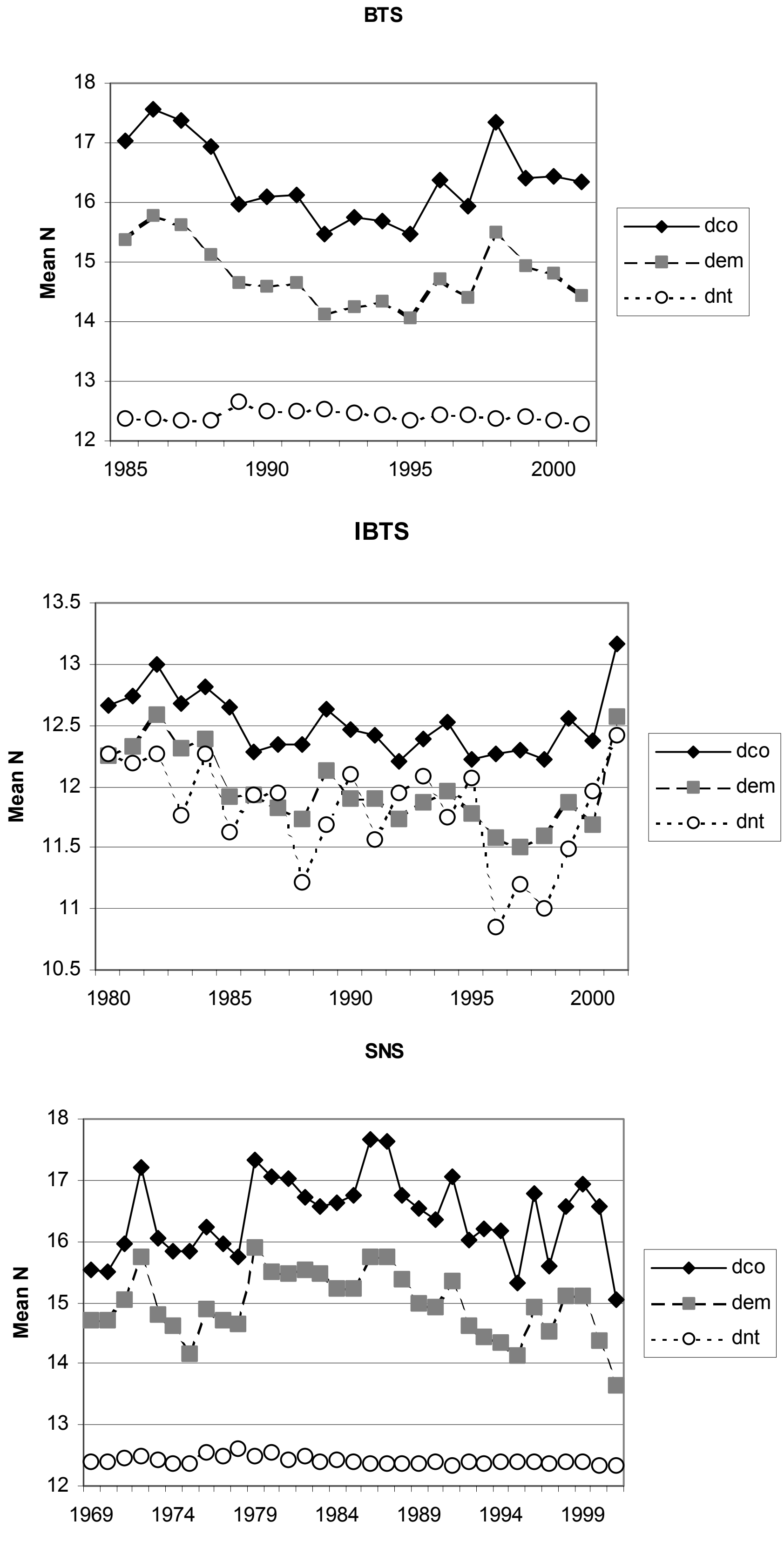


Figure 5.3.2.2.1.3. Time series of trophic level expressed by mean $\delta^{15}N$ in three surveys for different subsets of the fish community: dem=demersal, dco=demersal commercial and dnt=demersal non-target.



The significant decline in trophic level observed in the demersal fish community as sampled by the IBTS applied for both the commercial species and the non-target species. The other surveys (BTS and SNS) did not show significant trends except for the non-target species in the SNS (Figure 5.3.2.2.1.3).

5.3.2.2.2 Trends in biomass size spectra

The slopes and intercepts of the biomass size-spectra, describing both the entire community and the demersal community (Figure. 5.3.2.2.2.1), increased significantly from 1982-2000 (Sens tests, p<0.05). The mean $\log_2$ body mass and mean maximum $\log_2$ body mass of the demersal community fell significantly in the same period, while there was no significant change in the mean $\log_2$ body mass and mean maximum $\log_2$ body mass of the whole community (Figure. 5.3.2.2.2.2). In the SGFS the slopes and intercepts of the biomass size-spectra increased significantly from 1925-1996 (Figure. 5.3.2.2.2.3).

Figure 5.3.2.2.2.1. Slopes and intercepts of biomass size spectra (relationships between $\log_{10}$ normalised biomass by $\log_2$ body mass and $\log_2$ body mass) for the fish community sampled by the IBTS.

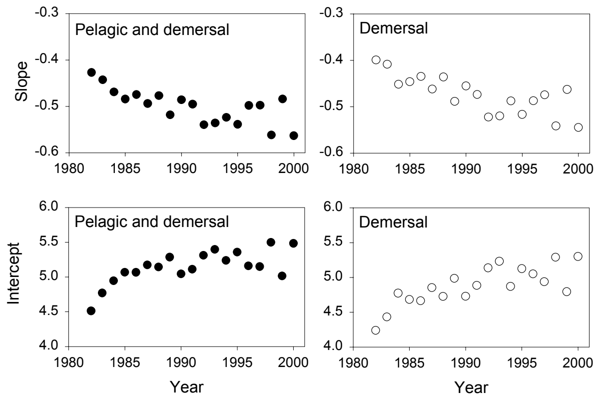


Figure 5.3.2.2.2.3. Slopes and intercepts of biomass size spectra (relationships between $\log_{10}$ normalised biomass by $\log_2$ body mass vs. $\log_2$ body mass) for the demersal fish community sampled by the SGFS.

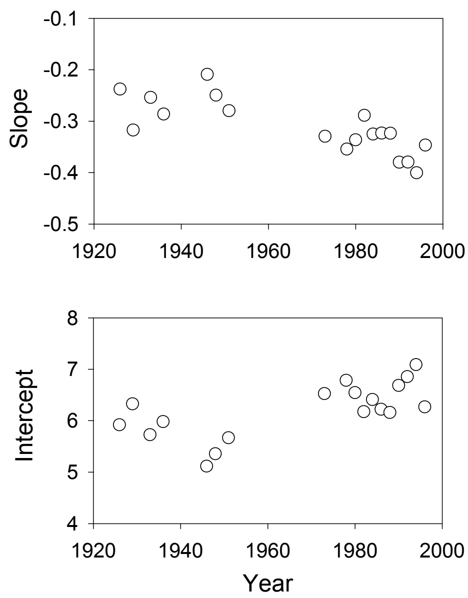
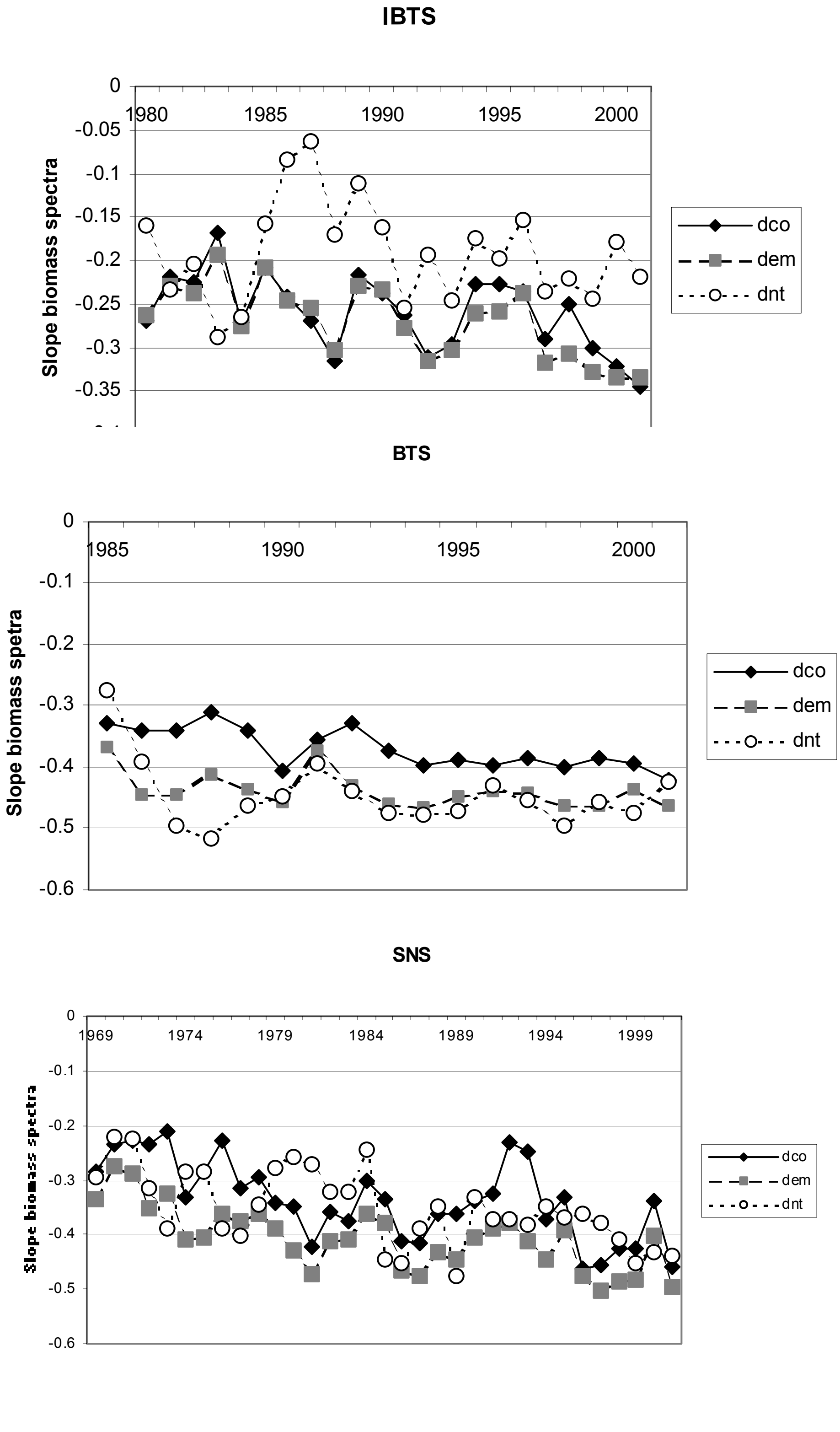


Figure 5.3.2.2.2.2. Trends in slope of biomass size spectra in three surveys for different subsets of the fish community: dem=demersal, dco=demersal commercial and dnt=demersal non-target.



The commercial species do not show a trend in the slope of their biomass size spectra in any of the surveys (IBTS, BTS, SNS). Increasing trends are observed for the non-target species in the IBTS and SNS.

5.3.2.2.3 Trends in mean weight and mean maximum weight

The mean $\log_2$ body mass and mean maximum $\log_2$ body mass of the demersal community showed a significant decline in both the IBTS and the SGFS (Figure 5.3.2.2.3.1).

Figure 5.3.2.2.3.1. Long-term trends in the mean $\log_2$ body mass and mean maximum $\log_2$ body mass of all species and of demersal species in the community sampled by the IBTS.

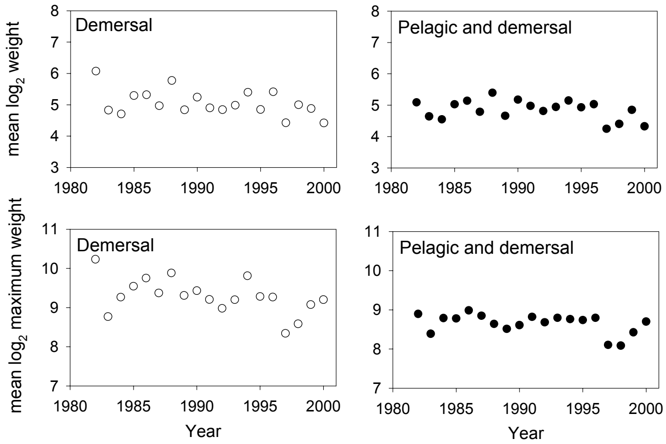


Figure 5.3.2.2.3.2. Long-term trends in the mean $\log_2$ body mass and mean maximum $\log_2$ body mass of demersal species sampled by the SGFS.

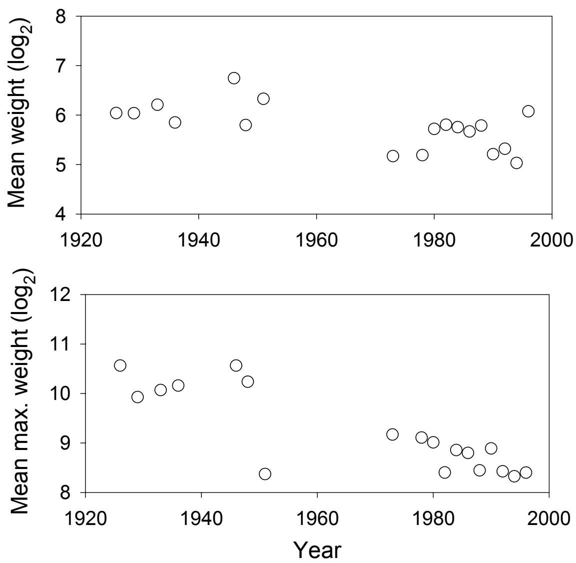
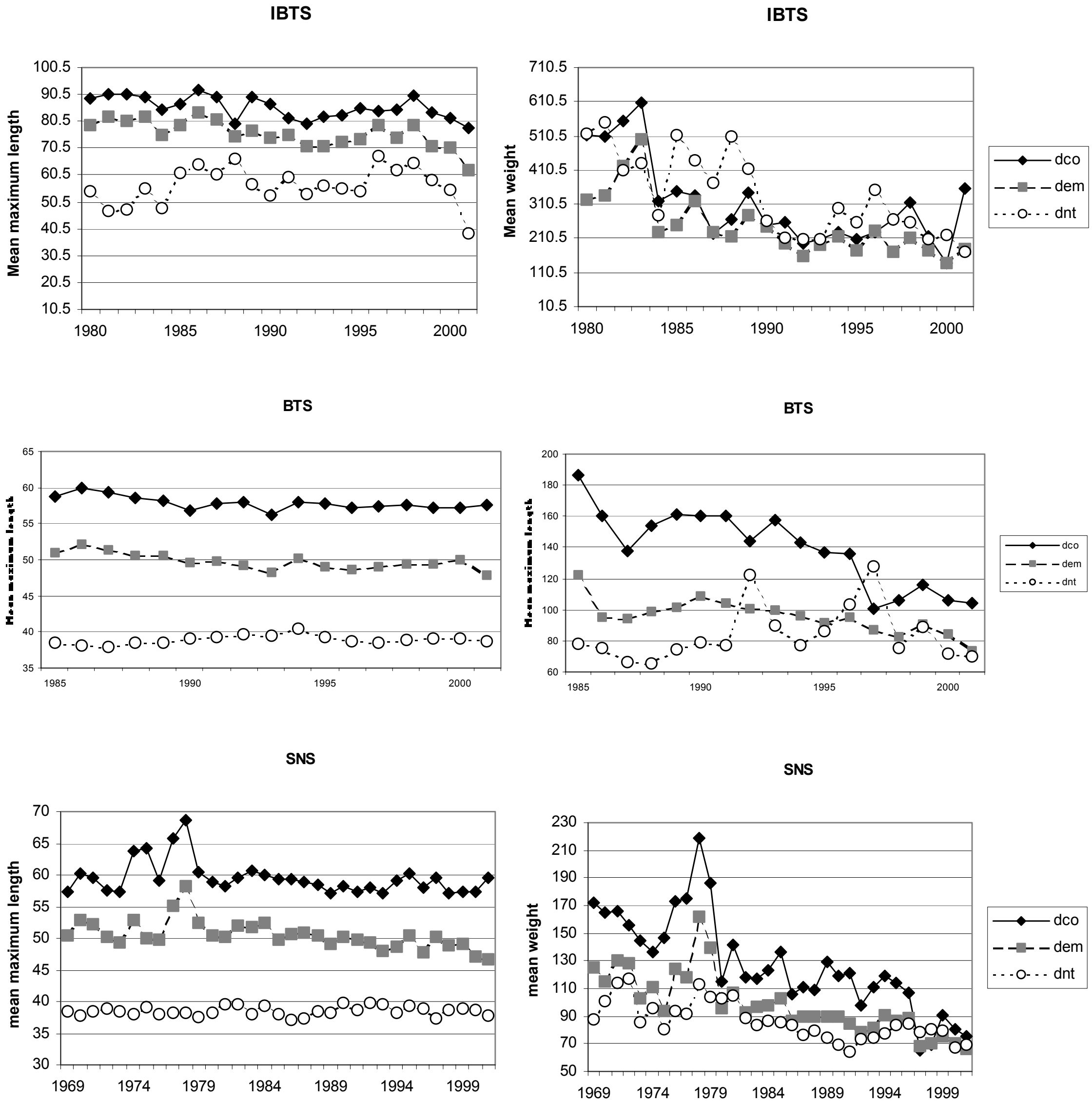


Figure 5.3.2.2.3.3. Time series of mean maximum length and mean weight in three surveys for different subsets of the fish community: dem=demersal, dco=demersal commercial and dnt=demersal non-target.

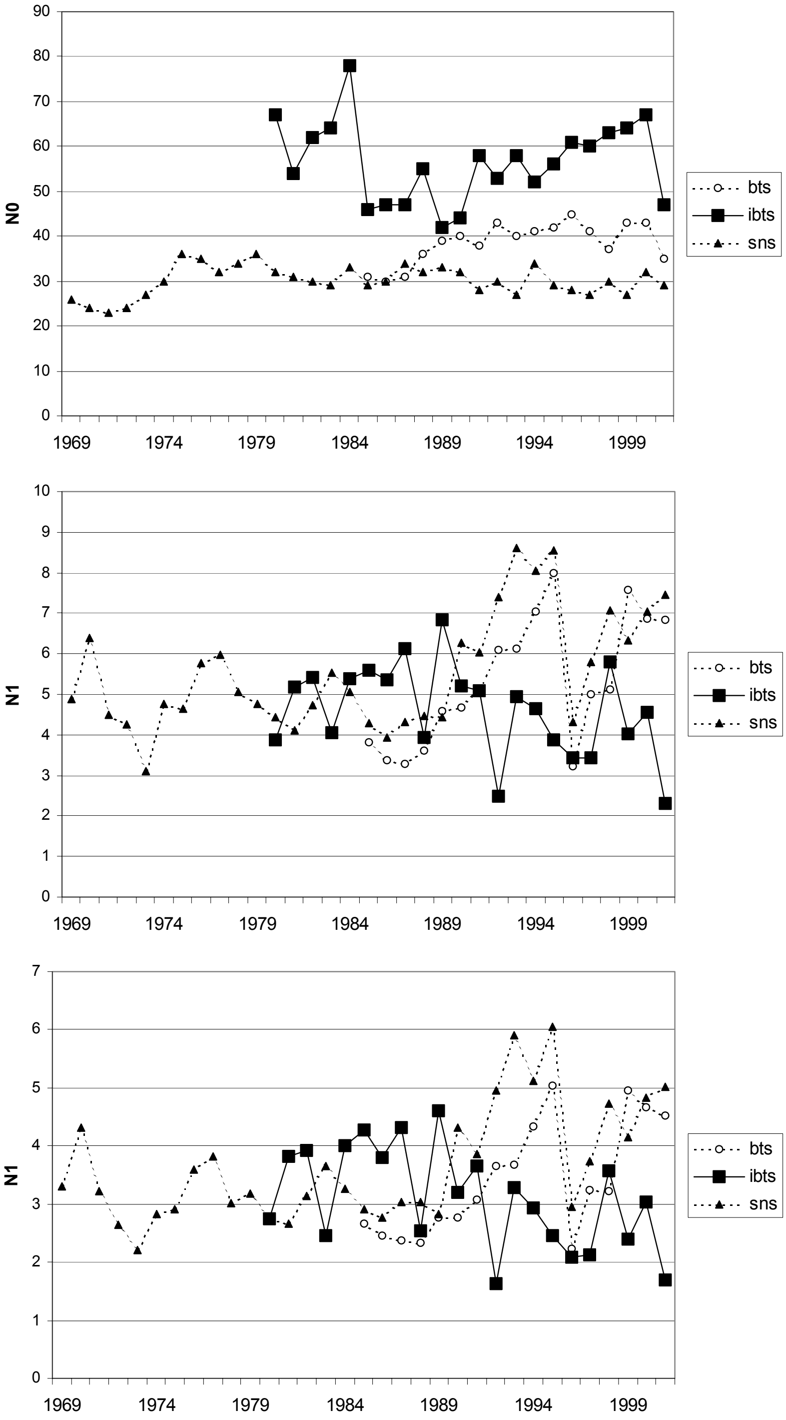


Mean weight of both the commercial and non-target species decreased significantly in all surveys (IBTS, BTS and SNS). This was only the case for the mean maximum length of the commercial species, not of the non-target species.

5.3.2.2.4 Trends in biodiversity indices

Significantly increasing trends are observed for the demersal fish assemblage in Hill's N0 in the BTS and Hill's N1 in the BTS and SNS. Remarkable were the inverse trends for N2 in BTS and SNS versus IBTS.

Figure 5.3.2.2.4.1. Time series of biodiversity indices over time in three surveys.



5.3.2.2.5 Comparison of trends among metrics and surveys

The correlations between mean trophic level, the slopes and intercepts of size spectra, and mean $\log_2$ body mass and mean maximum $\log_2$ body mass (Table 5.3.2.2.5.1) show that the size-based metrics of community structure are not consistently informative about the trophic structure of the same community. For all species sampled by the IBTS, mean $\delta^{15}N$ was weakly correlated with the slopes and intercepts of the size-spectra, but the correlations with the other size-based metrics were weak and opposing. For the demersal community sampled on the IBTS, the correlations between metrics were usually stronger than for the whole community with an increasing slope in the size spectra and decreasing mean or mean maximum $\log_2$ body mass reflecting the decline in trophic level. For the demersal community sampled by the SGFS, the changes in weight-based metrics and size-spectra are correlated, but the correlations between $\delta^{15}N$ and these metrics are weak and in opposing directions.

Table 5.3.2.2.5.1. Correlations between mean $\delta^{15}N$, mean $\log_2$ weight, mean maximum $\log_2$ weight and the slopes and intercepts of size spectra (plots of $\log_{10}$ normalised catch by $\log_2$ size class vs. $\log_2$ size class).

International Bottom Trawl Survey (IBTS) All species			
	mean $\delta^{15}N$	**mean $\log_2$ weight**	**mean max. $\log_2$ weight**
mean $\delta^{15}N$	-	-	-
mean $\log_2$ weight	−0.35	-	-
mean max. $\log_2$ weight	0.06	0.61	-
slope	0.35	0.36	0.22
intercept	−0.49	−0.29	−0.21

International Bottom Trawl Survey (IBTS) Demersal species			
	mean $\delta^{15}N$	**mean $\log_2$ weight**	**mean max. $\log_2$ weight**
mean $\delta^{15}N$	-	-	-
mean $\log_2$ weight	0.53	-	-
mean max. $\log_2$ weight	0.61	0.80	-
Slope	0.51	0.52	0.45
intercept	−0.55	−0.56	−0.49

Scottish Groundfish Survey (SGFS) Demersal species			
	mean $\delta^{15}N$	**mean $\log_2$ weight**	**mean max. $\log_2$ weight**
mean $\delta^{15}N$	-	-	-
mean $\log_2$ weight	0.12	-	-
mean max. $\log_2$ weight	−0.42	0.47	-
slope	−0.19	0.81	0.74
intercept	0.02	−0.85	−0.62

Table 5.3.2.2.5.2 allows comparison of trends between metrics and surveys. The tables show that for a particular metric there is never any contradiction between the surveys although in some cases not all of them show a significant trend. Also, the trends observed for the different selections of species are consistent. Over time the demersal fish assemblage and subsets show an increase in the slope of the biomass size spectra whereas mean weight, mean maximum length, and trophic level decreased. The metric that shows the strongest signal is that of the mean weight. This metric shows for practically every survey/species subset combination (except for BTS, non-target species) a significant downward trend.

Table 5.3.2.2.5.2. Trends observed in three surveys and four metrics for the demersal fish assemblage (top) and two subsets of this assemblage: the commercial species (middle) and the non-target species (bottom). Explanation codes: 1=positive significant (p<0.05), 0 not significant, −1 negative significant (p<0.05).

Demersal species				
Survey	**Slope Biomass spectra**	**Mean weight**	**Mean maximum length**	**Trophic level**
BTS	1	−1	−1	0
IBTS	0	−1	−1	−1
SNS	1	−1	−1	0

Demersal Commercial species				
Survey	**Slope Biomass spectra**	**Mean weight**	**Mean maximum length**	**Trophic level**
BTS	0	−1	−1	0
IBTS	0	−1	−1	−1
SNS	0	−1	−1	0

Demersal non-target species				
Survey	**Slope Biomass spectra**	**Mean weight**	**Mean maximum length**	**Trophic level**
BTS	0	0	0	0
IBTS	1	−1	0	−1
SNS	1	−1	0	−1

Table 5.3.2.2.5.3. Trends in biodiversity indices observed in three surveys for the demersal fish assemblage. Explanation codes: 1=positive significant (p<0.05), 0 not significant, −1 negative significant (p<0.05).

Demersal species			
Survey	**N0**	**N1**	**N2**
BTS	1	1	1
IBTS	0	0	−1
SNS	0	1	1

5.3.2.2.6 Assessing relationship between metrics and fishing effort

For the most sensitive metric the relationship with fishing effort was explored. Two approaches were used for this: (1) compare the pattern in areas where management measures resulted in marked changes in effort over time and (2) use the information on distribution of international effort of otter and beam trawling per ICES rectangle to distinguish between high and low effort areas and compare the pattern in these areas.

To reduce the discarding of plaice in the nursery grounds along the continental coast of the North Sea, an area between 53 ^{o}N and 57 ^{o}N was closed to fishing for trawlers with engine power of more than 300 hp in the second and third quarters since 1989, and for the whole year since 1995. These measures resulted in a marked shift of fishing effort to offshore areas. For the analysis two areas were distinguished: inside and outside the plaice box and three periods: 1985–1989 high effort inside the box, 1990–1994 medium effort, and 1995–2001 low effort. Linear regression was used to determine for each of these area-period combinations whether a trend was observed. For the BTS only the area outside the plaice box showed significant decreases in mean weight for periods when the plaice box was (partially) closed and effort redirected offshore. The trends of the commercial and non-target species in the box area observed in the SNS coincide with those observed in the BTS.

Table 5.3.2.2.6.1. Slope of mean weight in BTS survey conducted inside and outside the box area for three periods of time (period 1: 1985–1989, period 2: 1990–1994, period 3: 1995–2001)

Demersal species		
	Box	**Rest**
Period 1	−0.09	−0.09
Period 2	−0.22	−0.27
Period 3	−0.16	−0.18

Commercial species		
	Box	**Rest**
Period 1	−0.03	−0.05
Period 2	−0.13	−0.07
Period 3	−0.06	−0.10

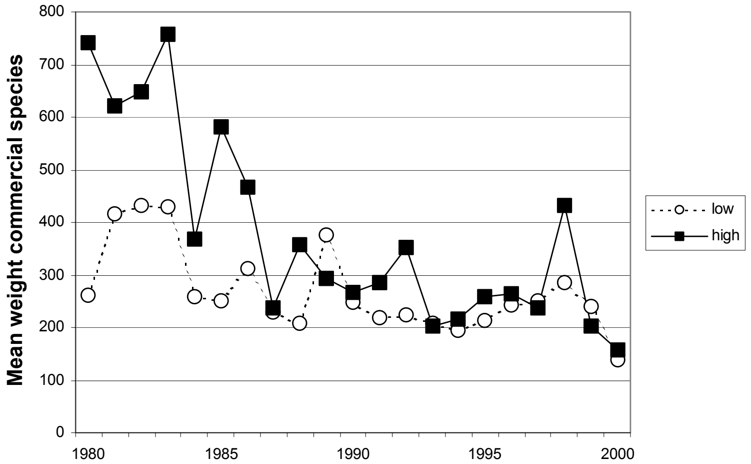
Non-target species		
	Box	**Rest**
Period 1	−0.10	−0.01
Period 2	0.23	0.00
Period 3	−0.11	−0.04

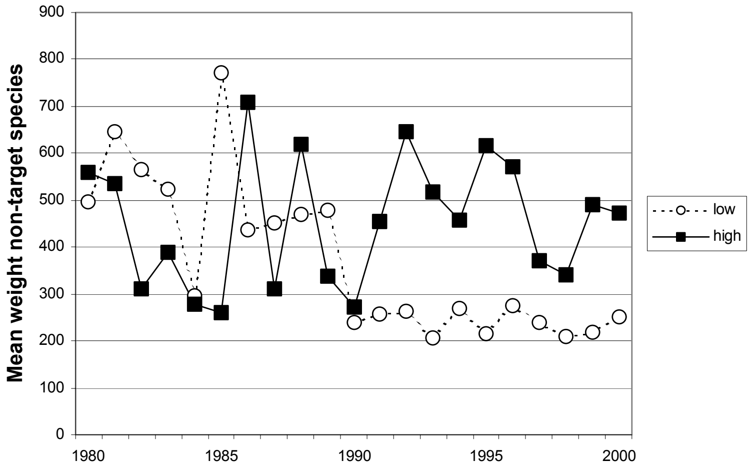
Table 5.3.2.2.6.2. Slope of mean weight in SNS survey conducted inside the box area for three periods of time (period 1: 1985–1989, period 2: 1990–1994, period 3: 1995–2001).

Period	**Assemblage**		
	dco	**dem**	**dnt**
1	−0.13	−0.18	−0.28
2	−0.03	0.01	0.25
3	−0.07	−0.19	−0.28

Based on the data describing the distribution in 1998 of international effort of otter and beam trawling per ICES rectangle (Jennings *et al.,* 1999a, 2000), two suites of rectangles or “treatments” were distinguished: one with low effort (< 10,000 fishing hours, 116 rectangles), another with high effort (>10,000 fishing hours, 105 rectangles). The assumption is that the distribution of effort over the years is consistent and that the data for 1998 are representative of this distribution.

Figure 5.3.2.2.6.1. Mean weight over time for commercial and non-target species in the IBTS in high effort and low effort rectangles.





In the early 1980s the commercial species show a strong decrease in mean weight in the high effort rectangles while the non-target species show a sudden decline around 1990 in the low effort rectangles. So, although changes between the two areas are apparent, they are not readily interpretable.

5.3.2.3 Discussion

Our analyses suggest that the trophic level of the whole North Sea fish community has decreased from 1982 to 2000. This matches an assumed increase in fishing intensity on the target demersal species. The decrease was significant only when we accounted for changes in the size structure of the community. Environmental change is unlikely to explain consistent patterns of change in size and trophic structure because there were long-term decreases in the abundance of all larger species and individuals and yet these species have a wide range of environmental preferences (Hempel, 1978; Jakobsson *et al.,* 1994; Daan and Richardson, 1996; Jennings *et al.,* 1999a; O'Brien *et al.,* 2000)

For the whole North Sea community (pelagic and demersal species), the slopes of the normalised biomass size spectra ($\log_2$ body mass classes) and the mean trophic level of the community were correlated, suggesting that these changes reflect changes in the trophic structure of the community (Rice and Gislason, 1996; Gislason and Rice, 1998; Bianchi *et al.,* 2000), and that increases in the slope of size spectra result in decreases in the mean trophic level of the community. Clearly, consistent relationships between slopes of size spectra and trophic structure of the community have considerable practical significance because they suggest that easily and cheaply measured size-based ecosystem metrics (e.g., Rice, 2000) could act as surrogates for complex descriptions of trophic structure. Analyses linking size and trophic structure could be further improved by the incorporation of data for invertebrates and small pelagic fish. These data may help to reduce the apparent variance in the relationship between the slope of the size-spectra and mean trophic level because (i) fish and invertebrate production are closely coupled in the North Sea food web; and (ii) the proportion of invertebrate biomass and production by body-mass class will change in space and time.

Changes in the trophic level of the North Sea fish community as sampled during the IBTS are relatively small in comparison with those reported elsewhere. For example, Pinnegar *et al.* (2002) demonstrated that the trophic level of the Celtic Sea community decreased by approximately 0.15 trophic level from 1982–2000, even though they took no account of changes in size composition with time. The relatively small changes in the North Sea may be a result of the long history of fishing (the whole North Sea was already trawled by 1900 while much of the Celtic Sea was not fished until the 1970s (Smith, 1994) and the intensive directed fisheries on species at low trophic level. In the Celtic Sea, much of the decline in trophic level with time was a function of the proliferation of boarfish *Capros aper*, a species of no commercial value. Conversely, in the North Sea, any proliferation of sprat *Sprattus sprattus*, herring *Clupea harengus*, and sandeel *Ammodytes* spp. in the whole fish community and Norway pout *Trisopterus esmarki* in the demersal community, might have been masked by the impact of directed fisheries (Gislason, 1994).

Few species account for most of the biomass in the North Sea fish community (Sparholt, 1990), and this is reflected in the composition of survey catches. Thus, haddock *Melanogrammus aeglefinus*, whiting *Merlangius merlangus,* and Norway pout dominate the demersal community and these species, plus herring (and sandeel that were not well sampled by the GOV trawl), dominate the biomass of the entire community. In practice, small changes in the biomass and size structure of the abundant species have the most significant impacts on the structure of the North Sea food web. It is notable that these species include those that feed at high as well as at low trophic levels. Whiting, for example, although not one of the largest species, feeds at a high trophic level. Our analysis of the relationships between body mass and $\delta^{15}N$ further supports the observation that cross-species relationships between body size and trophic level are weak, even though there is strong size-based trophic structuring in fish communites (Jennings *et al.,* 2001).

The GOV and Aberdeen otter trawls used in the IBTS and SGFS are size and species selective, and the composition of trawl samples does not represent the composition of the whole community (Doubleday and Rivard, 1981). The GOV trawl samples some pelagic species, and enabled us to look at trends in the abundance of most species contributing to the overall biomass. The Aberdeen otter trawl was only suitable for sampling the demersal community (Greenstreet and Hall, 1996; Greenstreet *et al.,* 1999a). Neither gear sampled sandeels *Ammodytes* spp. effectively, and this prevented us from looking for evidence of the proliferation of a highly abundant and productive species that feeds at low trophic level.

Our treatment of the fish community as an isolated entity was unavoidable because time series species-size-abundance data for invertebrates are not available at equivalent scales. However, given the large contribution of invertebrates to biomass and production, changes in their abundance, caused by the direct and indirect effects of fishing, might have a considerable influence on the trophic level of fish species in the North Sea ecosystem. This is an important consideration when assessing the effects of trends in trophic level on energy cycling, because energy that cannot be processed by the fish community may fuel invertebrate production. It would be useful to conduct an analysis of trends in trophic level that incorporates all the faunal elements contributing to biomass and production within given body-size classes. However, the accurate integration of data from different sampling programmes is not feasible at present.

In examining the trophic structure of the demersal communties sampled during the IBTS and SGFS, neither data set suggested that the trophic level of the community has changed significantly during a period of increasing fishing intensity. However, the responses of the body size metrics (mean $\log_2$ body mass and mean maximum $\log_2$ body mass) and the slopes of the size spectra were consistent with long-term changes in community structure caused by fishing. Previous studies have shown that changes in species composition due to fishing led to an increase in mean growth rate of the community, while mean maximum size, age at maturity and size at maturity decreased. A phylogenetically-based analysis demonstrated that trends in community structure could be predicted from the differential responses of related species to fishing. Thus, species that decreased in abundance relative to their nearest relative matured later at a greater size, grew more slowly towards a greater maximum size, and had lower rates of potential population increase (Jennings *et al.,* 1999a). Such differential responses of species to fishing, coupled with the decrease in mean size of individuals within exploited populations (Beverton and Holt, 1957), would explain a steeper slope of the size-spectra. However, although the development of size-spectra is founded in the study of food web dynamics (e.g., Dickie *et al.,* 1987), size spectra for a selected component of the ecosystem may simply reflect differential responses to mortality rather than changes in the food web. Given that cross-species relationships between body size and trophic level for demersal species are weak or non significant (Jennings *et al.,* 2001), we cannot draw reliable conclusions about changes in the trophic structure of fish communities from changes in size spectra unless pelagic species are reasonably well sampled by the survey gear.

The use of relationships between body mass and $\delta^{15}N$ to estimate long-term trends in the trophic level of the fish community is based on a number of assumptions: (i) that $\delta^{15}N$ is linearly and positively related to trophic level, and that the fractionation of $\delta^{15}N$ with trophic level is equivalent in different species; (ii) that trophic level can be estimated from $\delta^{15}N$; (iii) that the relationship is time-invariant; and (iv) that estimated values of $\delta^{15}N$ for individuals are representative of the community as a whole. In relation to (i), we assumed that $\delta^{15}N$ increased by 3.4 ‰ per trophic level in all species (Minawaga and Wada, 1984), representing the mean of reported values between 1.3 ‰ and 5.3 ‰. This has prompted calls for experimental evaluation and validation (Adams and Sterner 2000). Such validation has not been completed, and is probably not feasible, for all the species in a complex open-sea food web, although an ongoing study in which bass *Dicentrarchus labrax* have been fed on sandeel *Ammodytes marinus* and dab *Limanda limanda* diets at a range of ambient North Sea temperatures has shown that mean fractionation between the predators and their prey on both diets was 3.4 ‰ (Chris Sweeting, pers com., University of Newcastle). These experiments provide support for the assumed enrichment, but are based on a situation where food was unlimited.

Assumptions (iii) and (iv) cannot be verified. A simple comparison for 30 of the study species in 2000 and 2001 suggested that there was some short-term consistency in the body mass and $\delta^{15}N$ relationship, but this does not clarify whether the relationship may be applied to the same species in 1925! Body sizes, sensory capacity, energy requirements, mouth parts and swimming ability may be expected to have a key effect on feeding strategies, and these should be fixed by evolutionary processes that operate on much longer time-scales than 80 years. However, we cannot account for prey switching and functional responses to the abundance of prey on shorter time-scales, nor can we allow for the impacts of fishing on energy recycling. Moreover, it remains uncertain whether the $\delta^{15}N$ estimates for relatively small samples of fish are representative of North Sea populations as a whole. Because juveniles and adults of many species use different habitats, it was impossible to collect the full size range of individuals from a single region. Consequently, size-related differences in $\delta^{15}N$ will be confounded by spatial differences.

These assumptions are similar to those that apply to analyses based on trophic level estimates from diet data (Yang, 1982) or Ecopath models (Pauly *et al.,* 1998, 2001). However, analyses of long-term changes in trophic level based on $\delta^{15}N$ have some advantages, because dietary data are unlikely to provide an adequate assessment of trophic level for species that switch diet frequently, prey on species that are digested at different rates or have unidentifiable gut contents. Instead, $\delta^{15}N$ reflects the composition of assimilated food and integrates the impacts of short-term changes in diet (Polunin and Pinnegar, in press). The use of $\delta^{15}N$ as an index of trophic level allowed an evaluation of changes in community size structure on trophic level, an impact that was previously ignored.

The conclusion that the trophic level of the North Sea fish community has decreased (all species sampled) or remained stable (demersal species) depends on the assumption that the slope and intercept of the relationship between trophic level and body mass vary randomly around mean values over time. However, if slopes and intercepts are functions of fishing intensity, then attempts to assess the long-term effects of fishing from contemporary trophic level estimates will provide misleading results. The links between trophic size-spectra and fishing intensity require more investigation before we can draw firm conclusions about the long-term effects of fishing on the North Sea food web.

5.3.3 Scotian Shelf

5.3.3.1 Introduction

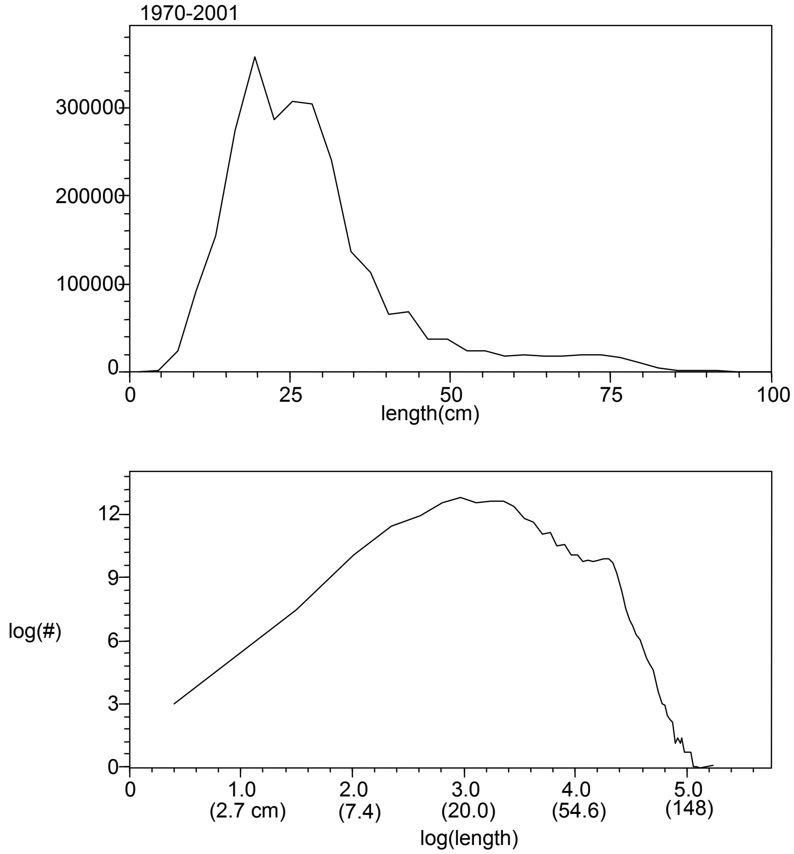
To provide spatial contrasts, the Scotian Shelf was divided in two manners. The first was approximately along the traditional management boundaries (the heavy line in Figure 5.2.3.1) of the western and eastern portions of the Shelf. The oceanography and stock structures of many of the dominant commercial species respects this division. The groundfishery in the eastern portion was closed in 1993 due to the collapse of the resident cod stocks. The second set of partitions was based on the distribution of effort (see Figure 5.2.3.2). The presumption is made that the effort distribution over the entire survey data period is similar to that seen in the 1990s, with the exception of the closure to groundfishing in the eastern area in 1993 and various area closures that were introduced from time to time.

As well as geographical decomposition of the data, three species subgroups were also compared: gadoids, elasmobranches, and flatfish.

5.3.3.2 Length-based data and analysis

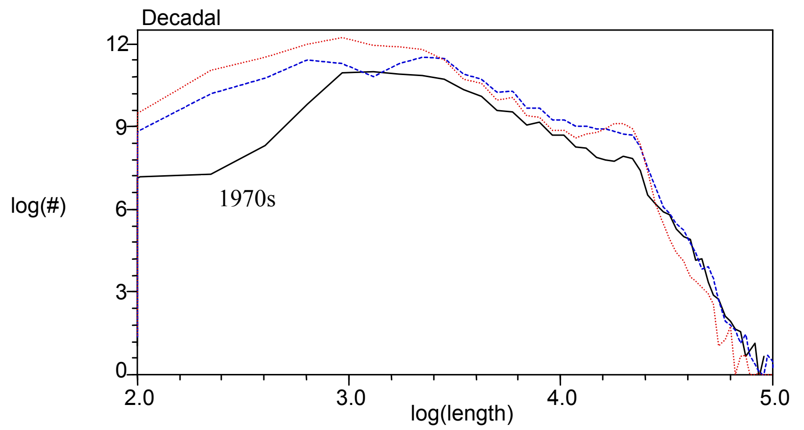
The length data from the summer RV survey underrepresents rare species. The protocol for taking lengths was focused on commercial species. Over 100 species were found in the abundance data but only 56 have lengths. An example of overall size spectra (1970–2001) for the Scotian Shelf is shown in Figure 5.3.3.2.1. The upper plot is the un-transformed data and is seen to peak at about 20 cm and then to fall rapidly until a plateau at 50 cm. The data are logged in the lower plot that also fall at two rates for sizes greater than 20 cm.

Figure 5.3.3.2.1. Size spectra for all Scotian Shelf areas, years (1970-2001) and species. The lower plot is the same data after taking the logarithm of both axes.



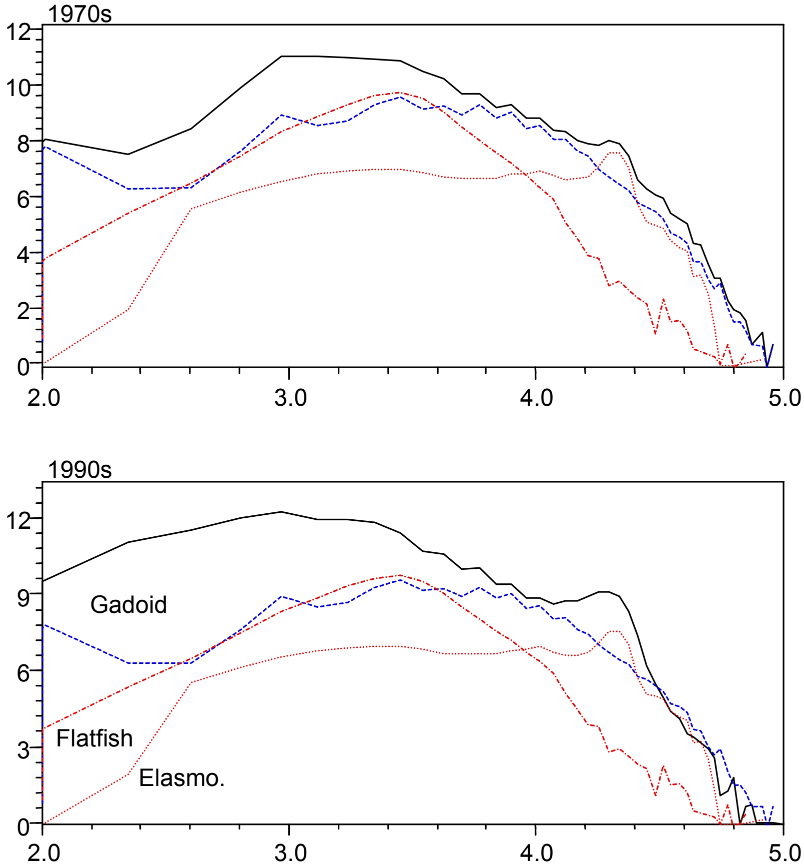
The data are separated into 3 decades to try and assess large-scale changes over the data period (Figure 5.3.3.2.2). The notch seen at log-length 4.3 is present in all three periods. The data for the period 1991–2000 show more small fish than the preceding decades. There is also a trend towards steeper slopes at larger sizes.

Figure 5.3.3.2.2. Log size spectra for all areas and species broken into 10-year periods. Solid line is 1971–1980, dashed line 1981–1990, and dotted line 1991–2000.



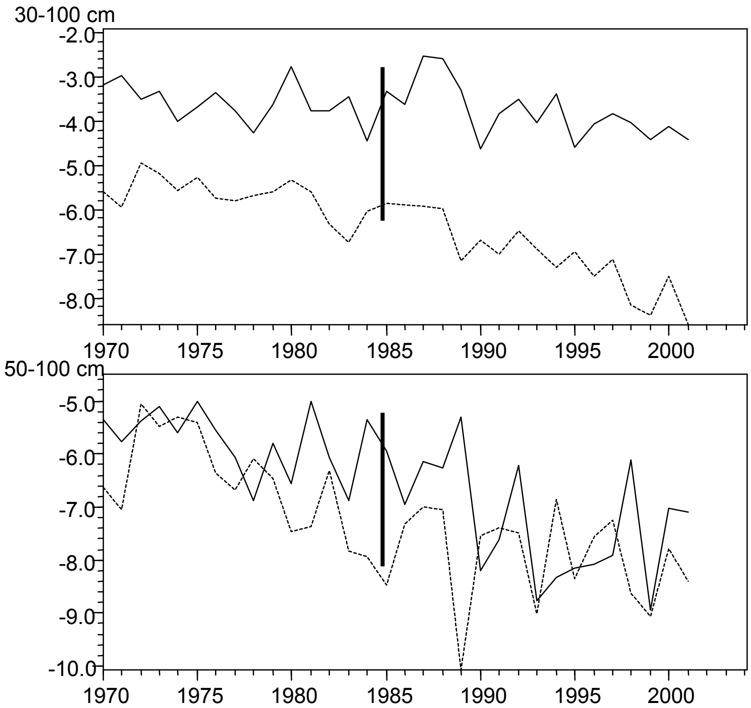
When each of the decadal spectra is decomposed into species groups (gadoids, elasmobranchs and flatfish), it is seen that the notch is due to elasmobranchs, which have become more prevalent over the data period (Figure 5.3.3.2.3).

Figure 5.3.3.2.3. Log size spectra for all areas and species broken into two 10-year periods (1970s and 1990s) and species groups. In each panel the gadoids are shown as dashed lines, the elasmobranchs as dotted lines and the flatfish as dash-dot lines.



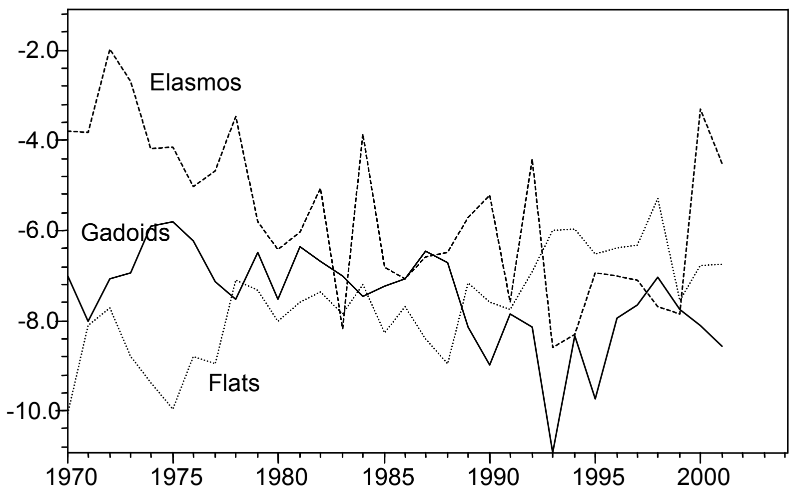
The slopes of the descending limb of the size spectra are often used as indicators of exploitation. They are dependent upon survivorship and in turn fishing mortality. If the growth rates change they also will affect the slope of the descending limb. The spectra are seen not to be linear, but have a domed-shape. Nonetheless, we fit a linear regression over a range of sizes after taking the logarithms. Figure 5.3.3.2.2.4 shows these slopes over two size ranges. The upper plot is from 30 cm to 100 cm and the lower limit was chosen to include fish well captured by the survey gear. The solid line is the western Scotian Shelf and is relatively constant over the data period. The slopes from the eastern Shelf show a steady decline, even after the closure of the fishery in 1993. The lower plot is from 50–100 cm on fish sizes that are exploited. The two are similar except in the 1980s when the eastern section gadoids were rapidly declining prior to closure.

Figure 5.3.3.2.4. Slope of log size spectra for all species for all western (solid line) and eastern (dashed line) areas. The solid vertical line marks the fishery closure on the eastern Shelf.



When the data are separated into species groups over the 50–100 cm range for the gadoids and elasmobranchs and 40–100 cm for the flatfish (Figure 5.3.3.2.5), they all show different trends over the data period. The slope for elasmobranchs falls over the first 15 years, while the gadoids are relatively constant. The flatfish show a general trend toward less steepness over the 30 years.

Figure 5.3.3.2.5. Slope of size spectra by species group (gadoids, elasmobranchs and flatfish).

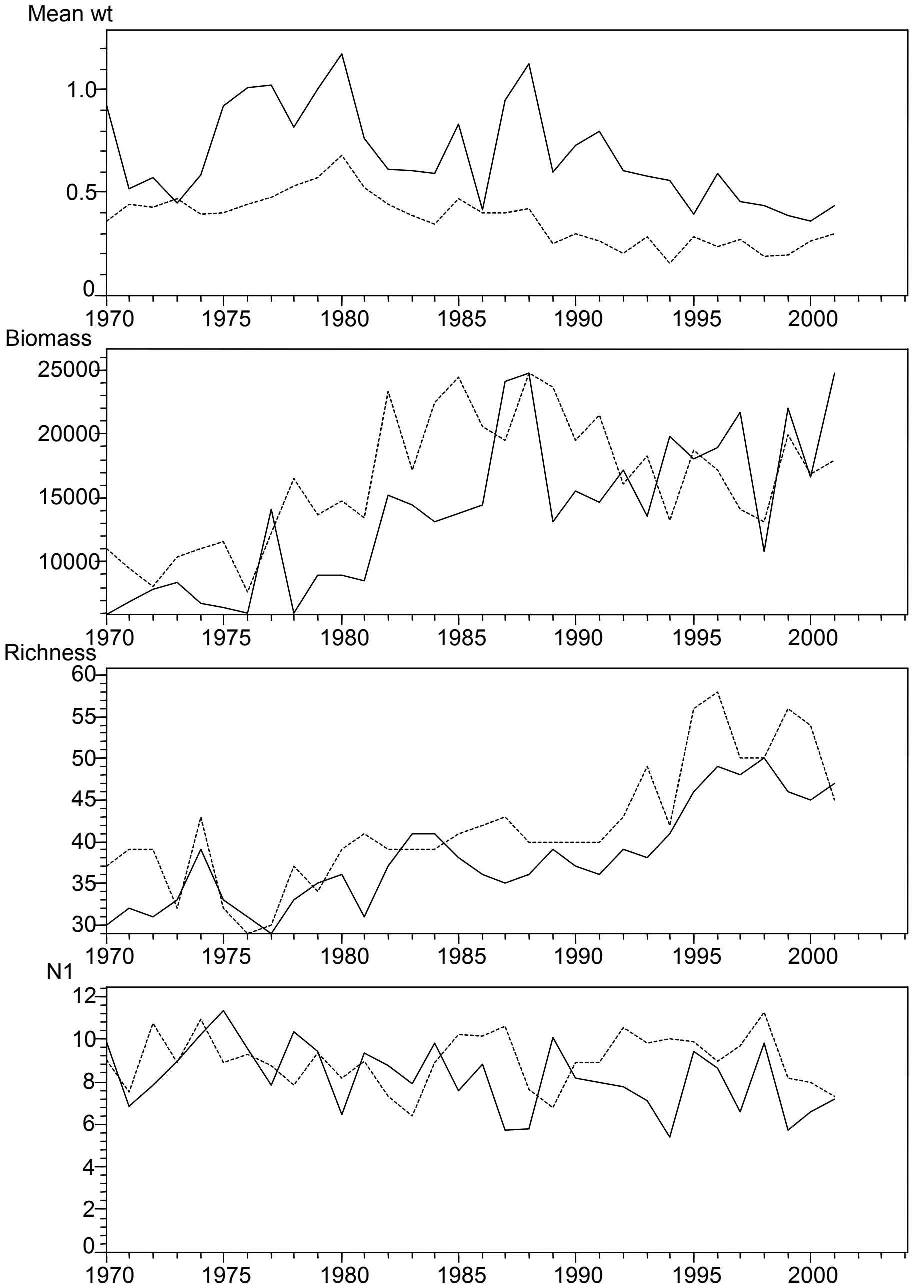


5.3.3.3 Community metrics

In this analysis five community indices are used for investigation of ecosystem response to fishing. Four of these are calculated directly from survey data: 1) the sum of weights of all species under consideration, and 2) the mean weight of organisms, richness and as a measure of diversity, Hill's N1. The fifth measure is the mean L_{Max} of species in a sample. For this exercise the L_{Max} was defined as the largest individual seen in the survey series. This is the only attribute to be averaged over the catch from these data and is used as a loose proxy for trophic level.

The mean weight of an animal caught (uppermost panel of Figure 5.3.3.3.1) for the Scotian Shelf shows a consistently higher weight for the western Shelf. Both areas show a declining trend since the early 1980s. The total biomass per tow (second panel, Figure 5.3.3.3.1.) shows similar trends in both areas with the eastern Shelf leading the increase in the early 1980s. The richness plot is the same for both areas and is aliased to some degree by changes in the sampling protocol, especially regarding invertebrate species. Finally, the diversity as estimated by Hill's N1 is stable throughout the data period and was insensitive to either the collapse of the fishery or the closure in the eastern area. Neither of the last two indices distinguish the eastern/western division of data. For these reasons, neither N1 nor richness will be used further in this study.

Figure 5.3.3.3.1. Community metrics for the eastern (dashed line) and western (solid line) Scotian Shelf.



The data were separated into the four effort zones described above for gadoids and elasmobranches (Figures 5.3.3.3.2 and 5.3.3.3.3). The effort zone breakdown of data for gadoids does not differentiate on an effort basis; the two western areas are similar even though they receive high and moderate effort. However, for elasmobranch data, the area of high effort shows a much higher biomass in recent years.

Figure 5.3.3.3.2. Gadoids by zone. The SSZ1 is the solid line, SSZ2 dashed line, SSZ3 dotted line and SSZ4 dash-dot line. The heavy vertical line denotes the closure of fishing on the eastern Shelf.

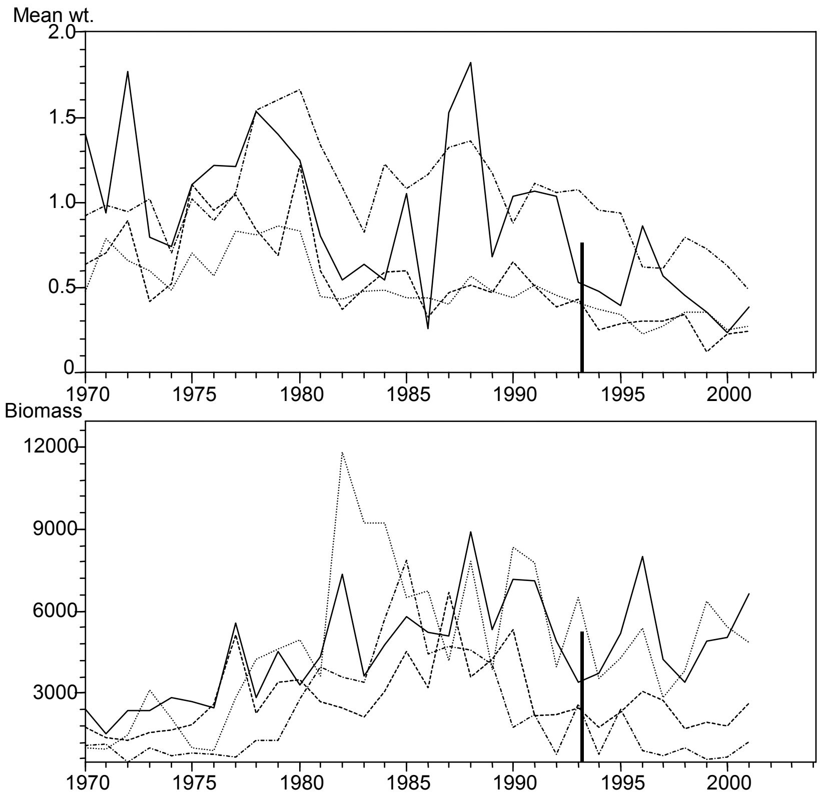
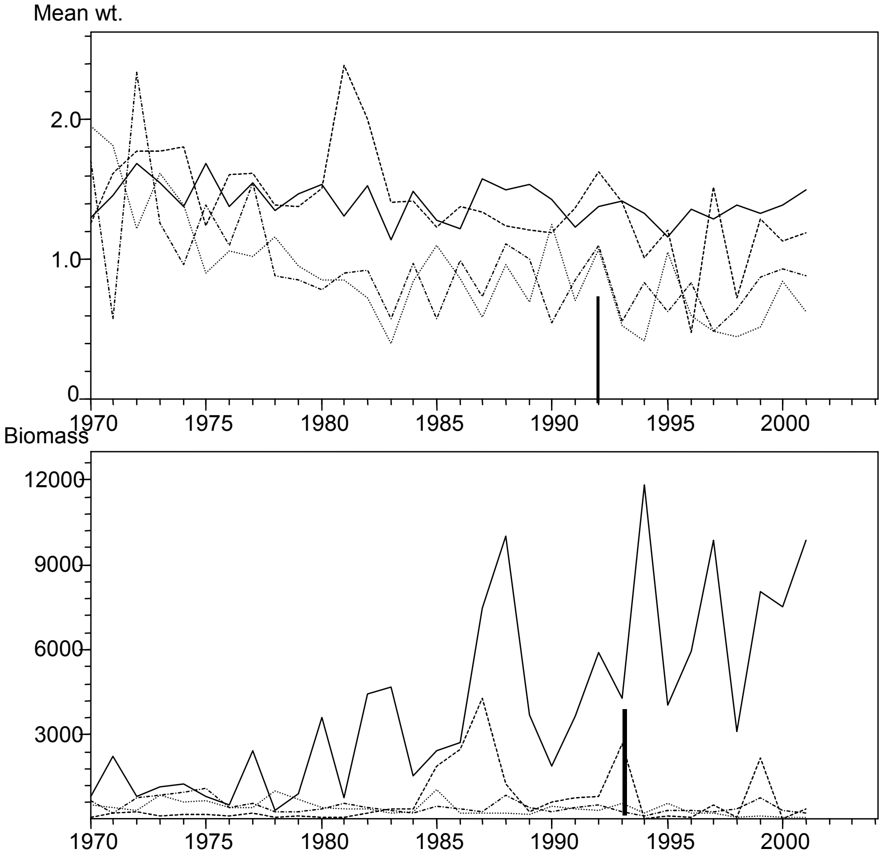
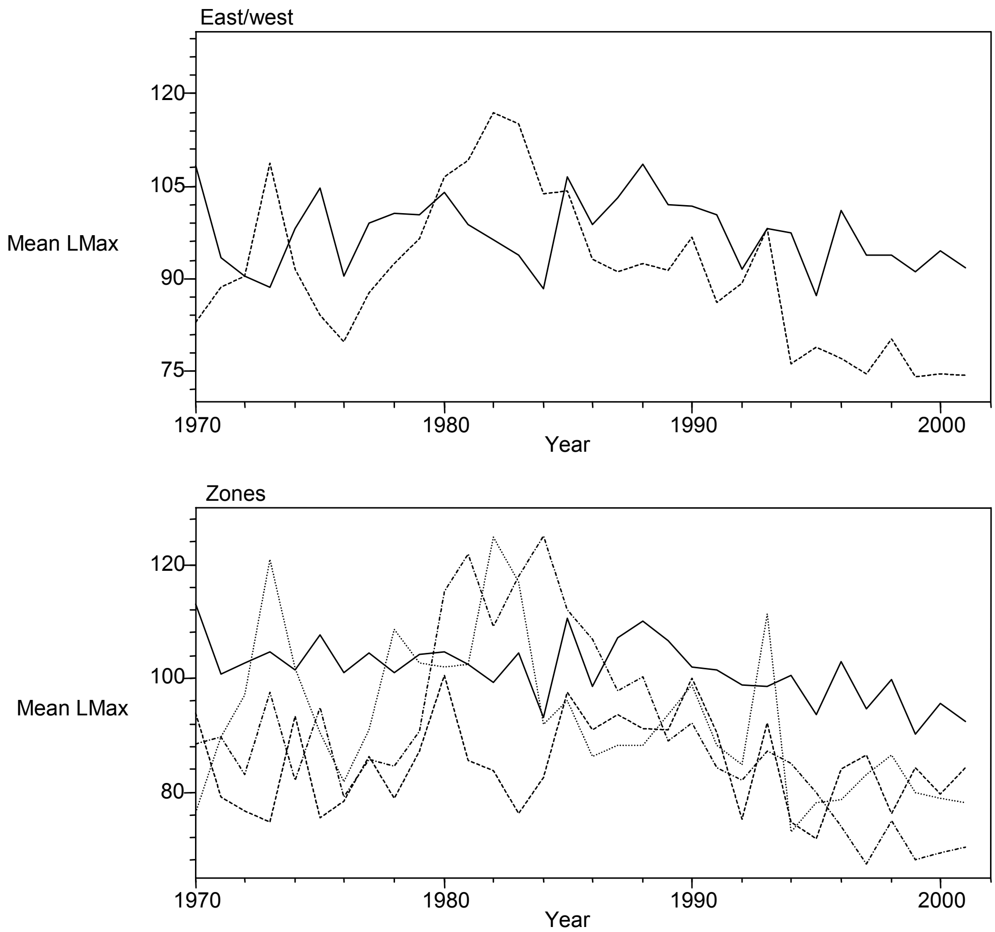


Figure 5.3.3.3.3. Elasmobranchs by zone. The SSZ1 is the solid line, SSZ2 dashed line, SSZ3 dotted line and SSZ4 dash-dot line. The heavy vertical line denotes the closure of fishing on the eastern Shelf.



The mean L_{max} is partitioned into areas and effort zones in Figure 5.3.3.3.4. The eastern area shows a strong decline from the early 1980s to the early 1990s. The timing is consistent with major events in the fishery of collapse and then closure. There is no apparent recovery since the closure in 1993. When the data are analysed by effort zone, the zone of highest effort (SSZ1) shows the greatest stability.

Figure 5.3.3.3.4. Mean trophic level by area (upper plot) and effort zone (lower plot) for all species. In the upper plot, the western area is shown as a solid line and eastern area as a dashed line. In the lower plot SSZ1 is the solid line, SSZ2 dashed line, SSZ3 dotted line and SSZ4 dash-dot line.



5.3.3.4 Conclusions

The Scotian Shelf analyses call into question the sensitivity of commonly used indices of community or ecosystem response to fishing, at least as revealed in RV survey data. The slope of the size structure is sensitive to the data window chosen and recruitment events can mask survivorship changes (Figure 5.3.3.2.4). When only fully recruited fish were used in the analysis, the two areas on the Shelf acted similarly and the eastern area remained constant after the fishery closure. When the size range was extended to pre-recruits, the slopes between the two areas were quite different and the index continued falling after the fishery closure. The continued fall is due to the appearance of more small (pre-recruit) fish which may be related to the removal of predators. The community metric analysis showed that N1 and richness did not respond to major events in the fishery nor did they discriminate major divisions of the Scotian Shelf. Biomass per tow and mean weight per tow were more sensitive. Even in terms of these two indices, the area of highest effort did not change over the data period, while the neighbouring areas of less effort did. The mean L_{max} weight did track major events but shows no response after the fishery closure. This could be either because the ecosystem failed to recover or because the index is insensitive.

5.3.4 Barents Sea

5.3.4.1 Introduction

5.3.4.1.1 Species composition

A list of fish species found in the Barents Sea includes 205 species (Dolgov, 2000). More precise distribution data are available for approximately 40 species. During the period 1993–2001, 91 fish species occurred in survey trawl catches. Most fish species (approximately 68 %) are widely distributed boreal species, while the remainder consists of Arctic species (32 %). 63 % of the species were associated with bottom or near-bottom habitats, while 32 % of the species were considered pelagic. Most of the species (51 %) are predatory fishes, while benthivorous and planktivorous species comprise 33 % and 15 %, respectively.

5.3.4.1.2 Dominant species

Despite the large number of fish species, only a few species are dominant in trawl catches. Nearly all species (99 % by number) come from a limited number of families (Gadidae, Pleuronectidae, Osmeridae and additionally Scorpaenidae, Clupeidae, Anarhichadidae, Rajidae, Cyclopteridae, Cottidae). On an annual basis only 10–12 fish species could be considered abundant, and 29-31 species represent 99 % of the trawl catches by number (Table 5.3.4.1.2.1). Abundances of all species found in the surveys were calculated as a number per 1-hour haul.

Table 5.3.4.1.2.1. List of the most abundant fish species from the Russian bottom surveys in the Barent Sea (1997–2001). All these (34) species are included in the analyses.

English name	*Latin name*
Northern wolffish	*Anarhichas denticulatus*
Atlantic wolffish	*Anarhichas lupus*
Spotted wolffish	*Anarhichas minor*
Greater argentine	*Argentina silus*
Atlantic hookear sculpin	*Artediellus atlanticus*
Polar cod	*Boreogadus saida*
Sea tadpole	*Careproctus reinhardti*
Atlantic herring	*Clupea harengus*
Polar sculpin	*Cottunculus microps*
Lumpsucker	*Cyclopterus lumpus*
Atlantic spiny lumpsucker	*Eumicrotremus spinosus*
Cod	*Gadus morhua*
Arctic staghorn sculpin	*Gymnacanthus tricuspis*
Long rough dab	*Hippoglossoides platessoides*
Atlantic poacher	*Leptagonus decagonus*
Dab	*Limanda limanda*
Esmark's eelpout	*Lycodes esmarki*
Vahl's eepout	*Lycodes vahli gracilis*
Onion-eye grenadier	*Macrourus berglax*
Capelin	*Mallotus villosus*
Haddock	*Melanogrammus aeglefinus*
Blue whiting	*Micromesistius poutassu*
Bull rout	*Myoxocephalus scorpius*
Plaice	*Pleuronectes platessa*
Saithe	*Pollachius virens*
Arctic skate	*Raja hyperborea*
Thorny skate	*Raja radiata*
Greenland halibut	*Reinhardtius hippoglossoides*
Golden redfish	*Sebastes marinus*
Deepwater redfish	*Sebastes mentella*
Norway haddock	*Sebastes viviparus*
Moustache sculpin	*Triglops murrayi*
Bigeye sculpin	*Triglops nybelini*
Norway pout	*Trisopterus esmarki*

5.3.4.1.3 Species characteristics

During the WGECO meeting the table with life history characteristics (L_{max}, L_{inf}, L_{mat}, A_{max}, A_{mat}, habitat, life style and trophic levels) for the Barents Sea fishes (205 species) was produced. All this information is given for the global distribution area of these species, so it is necessary to detail these data to the Barents Sea condition. Information about L_{max}, L_{mat}, A_{max} and A_{mat} should be available for at least 30–40 fish species from the Russian survey data.

5.3.4.2 Analyses and results

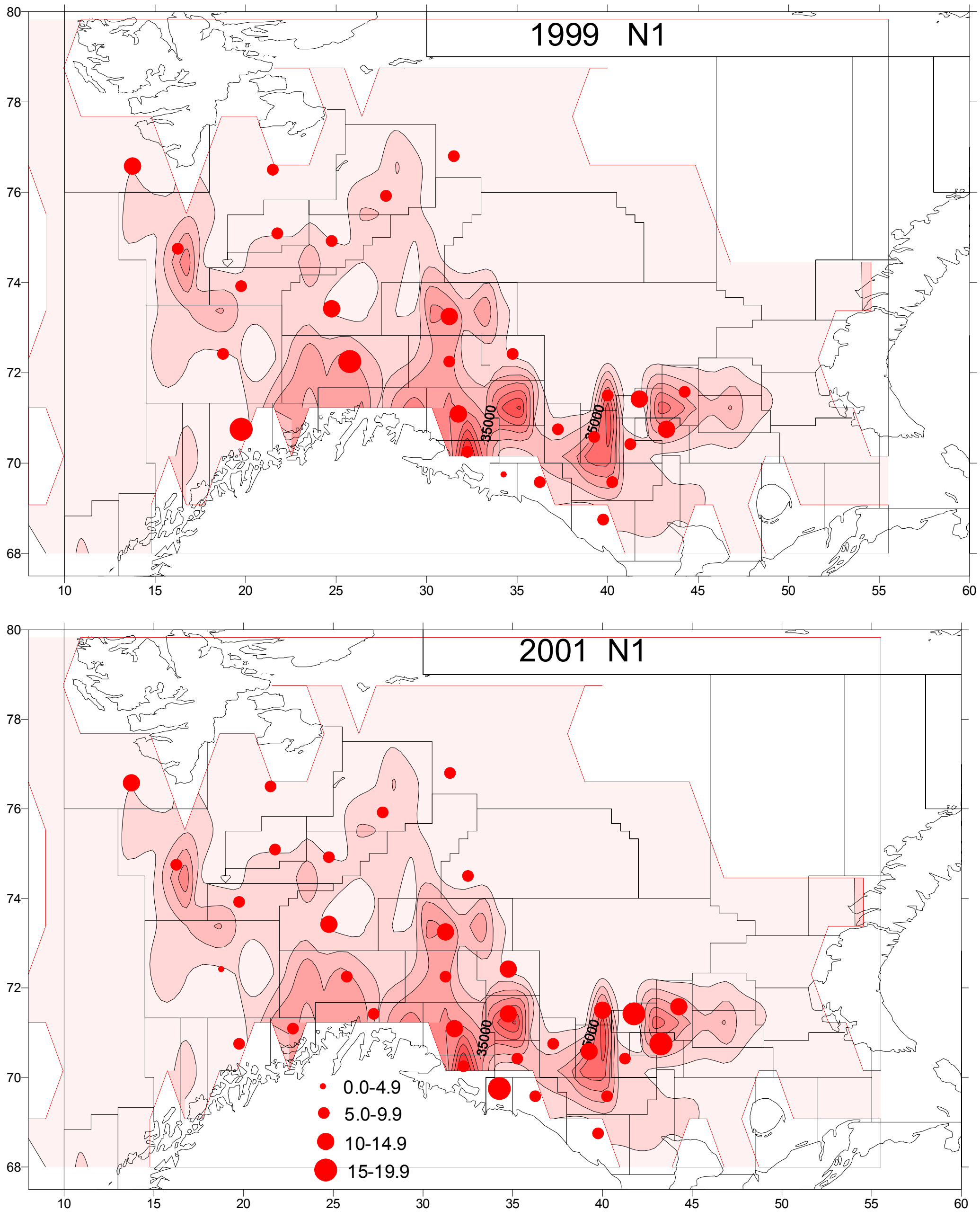
5.3.4.2.1 Biodiversity indices

Biodiversity indices were calculated for the 1999 and 2001 surveys – Hill's N0 and N1. Values were averaged by the local fisheries areas used in the Russian fisheries statistics for the Barents Sea. Significant spatial and temporal variations in biodiversity were shown (Figure 5.3.4.2.1.1). Species richness showed no obvious trends, e.g., northwards or eastwards, as could be expected because of the distribution of Arctic or boreal species.

An attempt was made to check one of the hypotheses that was tested at the last WGECO meeting (ICES, 2001): *Species richness should be lower in areas where current levels of fishing effort are highest, and should be higher in areas where recent levels of fishing effort are lowest.* However, both cases of high species richness and of low species richness in areas with intensive fisheries (Northern, Southern and Western slopes of Goose bank, Eastern Coastal area) were observed. It is necessary to note the increase of fish species numbers observed in surveys from 1999 to 2001 practically in all local areas. Only one case of decreasing species richness was observed for Nordkyn bank, which is an area with intensive fisheries. Therefore, this hypothesis does not seem to be supported by the presently available data for the Barents Sea fish community.

One possible explanation is the rather higher spatial and temporal differentiation of fish species composition in the Barents Sea in comparison, e.g., to the North Sea. Very extensive migrations for cod, capelin, polar cod, Greenland halibut and other fish species are known to occur. Furthermore, this is supported by data on environmental conditions, as there has been considerable warming in the western part of the sea and a northwards extension in the distribution of boreal species during the period studied.

Figure 5.3.4.2.1.1. Spatial distribution of Hill's indices N1 (circles) in 1999 and 2001 and commercial catches of cod (contour, combined for period 1984–1997) in various local fisheries areas.



5.3.4.2.2 Size spectra

Size spectra were calculated generally for the whole sea without any local differentiations (Figure 5.3.4.2.2.1). During 1997–2001 an increase of the intercept and slope was observed.

It was shown that values of the intercept and slope in the Barents Sea fish community are much lower than those in the North Sea and the Scotian Shelf. Figure 5.3.4.2.2.2 shows that the slope and intercept tend to decrease and increase compared to the North Sea and Scotian Shelf (Bianchi *et al.*, 2000).

Figure 5.3.4.2.2.1. The Barents Sea fish community size spectra from the Russian surveys in 1997–2001.

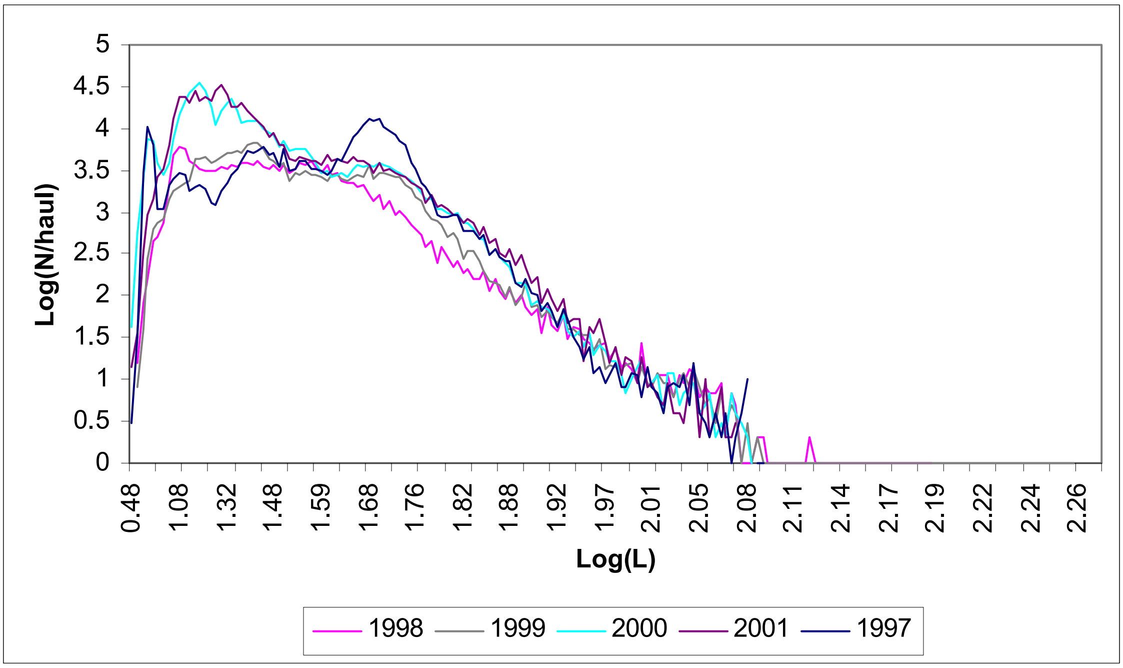
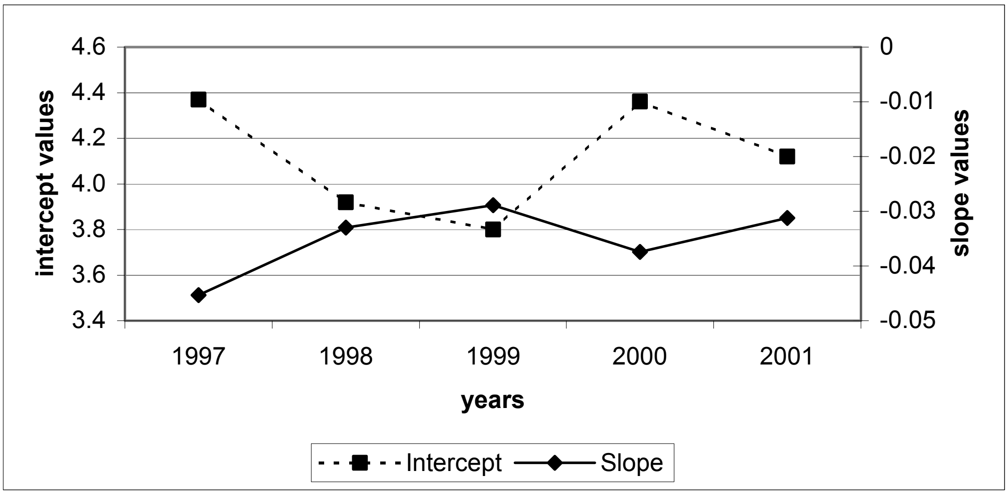


Figure 5.3.4.2.2.2. Values of intercept and slope of size spectra in 1997–2001.



5.3.4.3 Future work

This analysis is only preliminary, as the necessary data preparation required a lot of time. Further analysis will have to be carried out and should include the testing of all of the hypotheses which were tested for both the North Sea and

Portuguese waters in last year's report. This would enable comparisons to be made between fish communities in these waters.

For these objectives the following preparation needs to be done:

- Try to separate spatial assemblages taking into consideration biological and bathymetric factors as well as any other peculiarities of the Barents Sea;
- Control and check the information from surveys;
- Collect data on Russian trawl effort (both demersal, pelagic and shrimp fisheries), which should be combined by local areas;
- Detail all the life history characteristics of fishes for the Barents Sea.

5.3.5 Atlantic off the Portuguese coast

5.3.5.1 Introduction

These data were already analysed preliminarily during last year's WGECO meeting and in Bianchi *et al.* (2000). These analyses are therefore the continuation of this work and are complementary. However, because of the difficulty in acquiring effort data for this area, the number and types of analyses that can be carried out are limited.

Following the results of the analyses carried out last year (Hill *et al.,* 2001; ICES, 2001), a number of species were excluded from these analyses. These include *Capros aper* and *Macroramphous* spp. which are extremely abundant small migratory species that follow processes that are not fully understood, and which dominate the Portuguese assemblage (currently constituting 76 % and 8 %, respectively, of the entire assemblage), thus masking everything else. Also all species classified as pelagic in last year's "lifestyle" category were excluded. These species are listed in Table 5.3.5.1.1, and were excluded as these surveys are carried out using a bottom trawl, targeting demersal species, and so may not sample pelagic species thoroughly. Data for the remaining 173 species were used for the analyses.

Table 5.3.5.1.1. Species classified as pelagic and present in Portuguese surveys that were excluded from the following analyses.

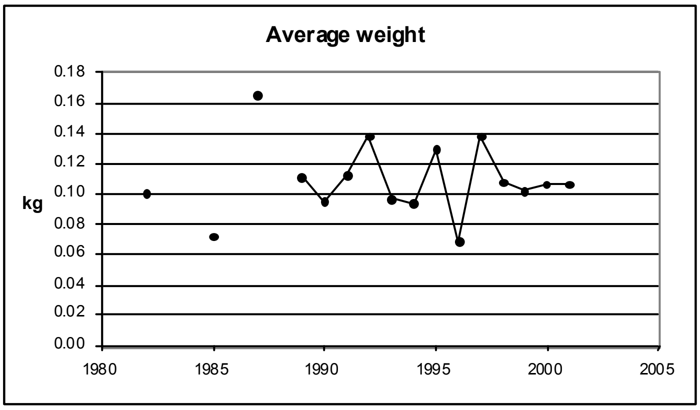
English Name	Latin Name
Allis shad	*Alosa alosa*
Twaite shad	*Alosa fallax*
Sand smelt	*Atherina presbyter*
Bullet tuna	*Auxis rochei*
Grey triggerfish	*Balistes carolinensis*
Garpike	*Belone belone*
Atlantic pomfret	*Brama brama*
Driftfish	*Cubiceps gracilis*
Anchovy	*Engraulis encrasicholus*
Shortfin mako	*Isurus oxyrinchus*
Golden grey mullet	*Liza aurata*
Thinlip mullet	*Liza ramada*
Blue whiting	*Micromesistius poutassou*
Sunfish	*Mola mola*
Flathead mullet	*Mugil cephalus*
Bluefish	*Pomatomus saltatrix*
Sardine	*Sardina pilchardus*
Chub mackerel	*Scomber japonicus*
Mackerel	*Scomber scombrus*
Atlantic saury	*Scomberesox saurus*
European sprat	*Sprattus sprattus*
Mediterranean horse mackerel	*Trachurus mediterraneus*
Jack mackerel	*Trachurus picturatus*
Horse mackerel	*Trachurus trachurus*
Swordfish	*Xiphias gladius*

5.3.5.2 Data analysis

5.3.5.2.1 Average fish weight in the whole assemblage

For this analysis the total biomass caught during a survey was divided by the total number of individuals in that survey, giving an annual average weight. This value was plotted against time (Figure 5.3.5.2.1.1).

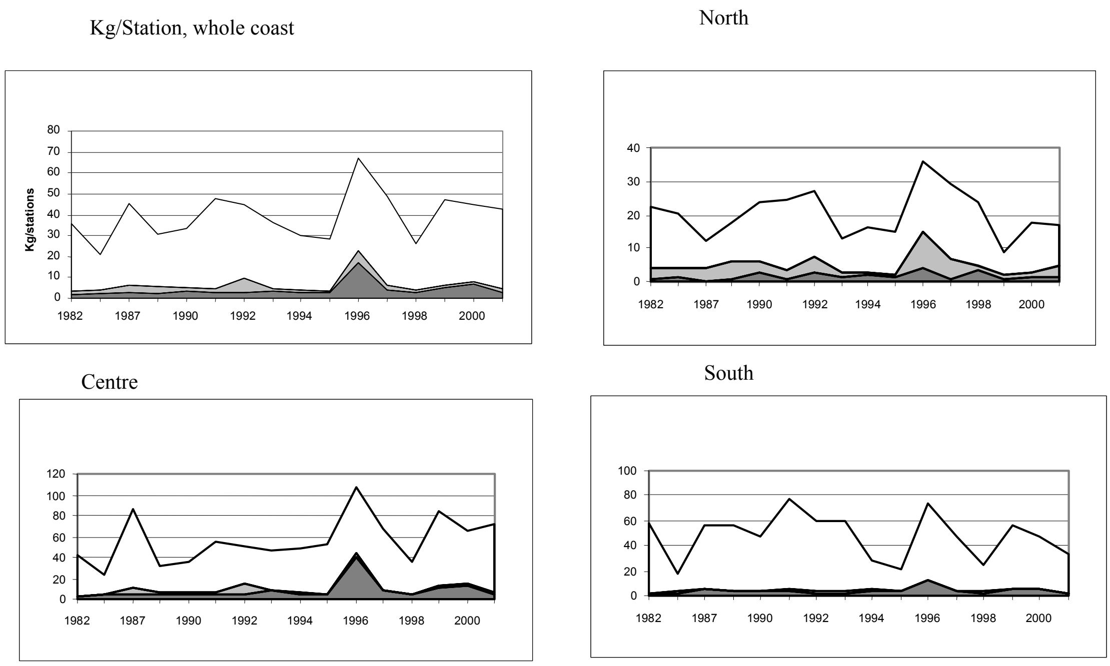
Figure 5.3.5.2.1.1. Average fish weight in the whole Portuguese assemblage.



5.3.5.2.2 Survey biomass divided into species groups

In order to assess how different species groups are evolving over time, each species was attributed to 1 of 3 species groups (elasmobranch, gadoid or other). These groups were chosen because of *a priori* assumptions about their life history characteristics, and because of the effects that fishing has been observed to have on them in other studies (Walker and Heessen, 1996; Walker and Hislop, 1998). The total weight of each species per station (corrected to 1 hour) was calculated and the sum of these totals weighted by the number of stations sampled that year to give an average weight of each species group. This was done both for the whole area and for three geographical regions (north, centre and south) to see whether the events observed were regional or general. The temporal evolution of these values was plotted as an area graph (Figure 5.3.5.2.2.1).

Figure 5.3.5.2.2.1. Area plot of the average biomass of gadoids (dark grey), elasmobranchs (light grey) and other species for the whole of the area and for the 3 geographic regions.

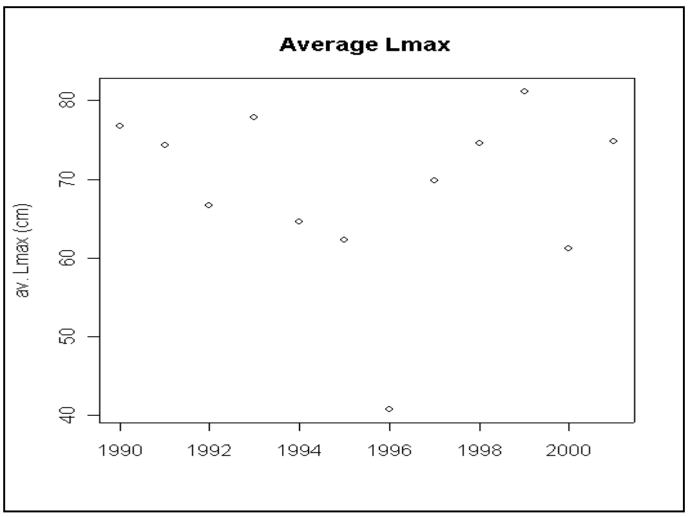


It can be seen that there was an “event” in the elasmobranch community in 1996 in the central region. This is caused by the lesser spotted dogfish (*Scyliorhinus canicula)*.

5.3.5.2.3 L_{max} analysis

For the length analysis this year, observed L_{max} per species was used, rather than the global values that were collected last year. This observed L_{max} is the largest individual of a given species recorded in all surveys. This overcomes the problem of unrealistic L_{max} values found when using global values. Furthermore, because of the problems identified in the discussion in Hill *et al.* (2001) due to the large number of small individuals of potentially large species (e.g., hake or horse mackerel) found close inshore in the third quarter due to recruits, it was decided to exclude all individuals smaller than 15 % of the observed L_{max}. This value was chosen based on ALKs for a few species, and because of the lack of ALKs for most species. The corrected average L_{max} was then plotted against time (Figure 5.3.5.2.3.1).

Figure 5.3.5.2.3.1. Average L_{max} for fish of the Portuguese coast for the period 1990 to 2001.



5.3.5.3 Conclusions

All these analyses back up the observations made last year (Hill *et al.,* 2001; ICES, 2001), that there are no apparent changes in the community make up of fish in the Atlantic ecosystem off the Portuguese coast during the time for which data are available. This is despite the fact that biological and environmental driving forces were taken into consideration (removal of pelagics, removal of small 0-year group fish).

5.4 Synthesis and Discussion

Text Box 1: Rationales that explain the impacts of fishing on community metrics.

Prediction 1: Species richness and species diversity of the groundfish assemblage are lower in areas most affected by fishing.

First-order effects: Fishing mortality is unequal (selective) among species and sizes. This selectivity results in a differential reduction in abundance of individual species. The degree of species selectivity of fishing, and the magnitude of reduction of the species experiencing greatest fishing mortality (whether as target species or by-catch) will determine whether diversity is increased or decreased. Where fishing mortality has been high and inflicted on many species, many once common species are much less abundant, and a small number of species are expected to increase in dominance in trawl catches. This effect will lead to a decrease in species diversity. If the fishing mortality is high enough to reduce some species to abundances so low that they are expected to be caught only rarely, then estimates of richness, as well as diversity, are expected to be reduced. If, on the other hand, fishing mainly targets the most abundant species, a high fishing mortality will result in an increased diversity.

Potential higher-order effects: In systems controlled by top-down predation, the reduction of predator abundance by fishing will result in a reduction in predation mortality, allowing prey species to increase in abundance. If the number of preferred prey is small relative to the number of exploited predators, or if the numerical response of prey to release from predation is large compared to the original predator abundances, the effect of release from predation would be expected to amplify the reduction in diversity through increasing dominance of the assemblage by a few prey species.

Prediction 2: The life history characteristics of the groundfish assemblage are affected by fishing – growth rates should be highest, and size at maturity, age at maturity, and ultimate body size should be lowest in areas most affected by fishing.

First-order effects: Fishing mortality has a direct effect on age-specific total mortality. When fishing mortality is a large percentage of total mortality, and is not uniform with all ages, the change in age-specific survivorship has many life history consequences. Many have already been discussed in Section 4 of last year's report, and some are discussed in Section 10 of this report.

Potential higher-order effects: Any change in life history characteristics has ecological consequences for the species and the assemblage in which it occurs. A large number of factors (top-down or bottom-up control, food web structure, frequency and magnitude of stochastic perturbations, etc.) may affect whether higher-order interactions amplify or buffer the direct effects of fishing on life history traits.

Prediction 3: The trophic level at which fish belonging to the groundfish assemblage feed is affected by fishing – trophic level should be lower in areas most disturbed by fishing.

First-order effects: Higher fish predators are size selective, and it was established by the Multispecies Assessment Working Group in the 1980s that the major North Sea fish predators had preferred predator-prey size ratios. Fishing necessarily makes larger individuals of exploited predator populations rarer. This, in turn, shifts the distribution of predation mortality to smaller prey. This would be expected to show up as a shift in predation mortality to lower trophic levels, as long as the assumption that predation is size dependent continues to hold.

Potential higher-order effects: Depending on the amount of cycling in the food web, taxa of similar sizes could potentially feed at quite different trophic levels, or even a given size of a particular taxon could feed at different trophic levels. Hence if fishing reduces predators relative to their prey, which in turn produces a change in the species composition of the prey or alters the role of cycling, the effects of fishing on predator trophic level could be buffered or amplified.

In the analyses on the North Sea, the analyses based on size-based metrics suggest that there are predictable community responses to fishing. These responses were reflected both in the slopes of size-spectra and by the simple metrics based on mean size and mean maximum body size. The response of the size spectra can reflect a number of first- and second-order fisheries impacts on the community, including: (1) the differential vulnerability of larger species; (2) within-population changes in mean body size and life history; (3) genetic changes in life history; and (4) predator-prey relationships in the community. Existing theoretical models that describe the structure of size spectra and their response to fishing focus on (1), (2) and (4). Mean size and mean maximum size of the community reflect (1)–(4) and (1), respectively. When presented together, these metrics will provide an indication of changes in both the size structure and species composition of the community (Piet, 2001).

Like the size-based metrics, the trophic metrics also show similar trends over time. In contrast, some of the biodiversity indices show inverse trends. In the northern North Sea, Hill's N1 (effectively the number of abundant species) and N2 (effectively the number of very abundant species) were lowest in areas with high fishing effort and the indices decreased over time (Section 5.3.1). The observed decline in Hill's N2 with time was supported by results of the IBTS that covers the entire North Sea (Section 5.3.2). In contrast, two surveys in the southeastern North Sea show an increase over time in Hill's N1 and N2 (Section 5.3.2). The observed inverse trends in different areas of the North Sea may be explained by the rationale in the text box and can be indicative of important differences between fish communities or fishing practices in these areas. In addition, the observed low diversity in areas with high fishing effort (Section 5.3.1) may be explained by the fishermen selecting areas with relatively high abundance of commercial species and thus low diversity instead of an effect of previous fishing impact. Cause and effect are not clear-cut in this case. In the Scotian Shelf data, N1 did not respond to major events in the fishery nor did it discriminate regional differences well.

Although the observed trends in Hill's N0 (species richness) are consistent among surveys, there are several concerns that apply to this metric. Species richness is highly dependent on sampling effort and species identification, and a recent analysis suggests that the IBTS database contains many species identification errors (Daan, 2001).

The size-based metrics we compared in all the ecosystems considered behaved differently. This may reflect the fundamental differences between the systems, especially in terms of the environment and history of fishing. Moreover, long time series of data (>30 years) were only available for the North Sea and the Scotian Shelf, and so the power to detect any change was lower in other systems. The slope of the size spectra tracked the decline of the resources in the eastern Scotian Shelf but has not responded since the fishery was closed in 1993, suggesting that either the ecosystem has not responded or that 8 years' data are insufficient to resolve a response. Comparisons of the size-based metrics for the Barents Sea and Portuguese waters were based on very short time series and did not reveal clear temporal trends.

Although the results in Section 5.3.1 suggest that a straightforward relationship exists between fishing effort and several of the community metrics, this was not confirmed by other analyses in the North Sea and the Scotian Shelf. Our results suggest that the interpretation of trends in both biodiversity and in length-based indices is not straightforward. Clearly more work, both empirical and theoretical, is needed to understand the impact of bottom fisheries on these indices.

References

Adams, T.S., and Sterner R.W. 2000. The effect of dietary nitrogen content on trophic level ^{15}N enrichment. Limnol Oceanogr, 45: 601–607.

Beverton, R.J.H., and Holt, S.J. 1957. On the dynamics of exploited fish populations. MAFF, London, UK.

Bianchi, G., Gislason, H., Graham, K., Hill, L., Jin, X., Koranteng, K., Manickchand-Heileman, S., Payar, I., Sainsbury, K., Sanchez, F., and Zwanenburg, K. 2000. Impact of fishing on size composition and diversity of demersal fish communities. ICES Journal of Marine Science, 57: 558–571.

Braber, L., and de Groot S.J. 1973. The food of five flatfish species. (Pleuronectiformes) in the southern North Sea. Neth J Sea Res 6: 163–172.

Cardador, F., Sanchéz, F., Pereiro, F.J., Borges, M.F., Caramelo, A.M., Azevedo, M., Silva, A., Pérez, 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, 30 pp.

Connell, J.H. 1975. Some mechanisms producing structure in natural communities. Ecology and Evolution of Communities (eds M.L. Cody & J.M. Diamond) Belknap Harvard University Press, Cambridge, Massachusetts, 460–490.

Connell, J.H. 1978. Diversity in tropical rainforests and coral reefs. Science, 199: 1302–1309.

Christensen, V. 1995. A model of trophic interactions in the North Sea in 1981, the year of the stomach. Dana, 11: 1–28.

Cushing, D.H. 1975. Marine ecology and fisheries. Cambridge University Press, Cambridge.

Daan, N., and Richardson, K. (Eds.) 1996. Changes in the North Sea ecosystem and their causes: Arhus 1975 revisited. ICES J Mar Sci 53: 879–1226.

Daan, N. 2001. A spatial and temporal diversity index taking into account species rarity, with an application to the North Sea fish community. ICES CM 2001/T:04. 6 pp.

Davidson, D.W. 1977. Species diversity and community organization in desert seed-eating ants. Ecology, 58: 711–724.

Dickie, L.M., Kerr, S.R., and Boudreau, P.R. 1987. Size-dependent processes underlying regularities in ecosystem structure. Ecol Monogr, 57: 233–250.

Dolgov, A.V. 2000 New data on composition and distribution of the Barents Sea ichthyofauna. ICES CM 2000/Mini:12.

Doubleday, W.G., and Rivard, D. 1981. Bottom trawl surveys. Can Spec Pub Fish Aquat Sci, 58: 273 pp.

France, R., Chandler, M., and Peters R. 1998. Mapping trophic continua of benthic food webs: body size $\delta^{15}N$ relationships. Mar Ecol Prog Ser, 174: 301–306.

Fry, B., and Quinones, R.B. 1994. Biomass spectra and stable-isotope indicators of trophic level in zooplankton of the northwest Atlantic. Mar Ecol Prog Ser, 112: 201–204.

Gislason, H. 1994. Ecosystem effects of fishing activities in the North Sea. Mar Poll Bull, 29: 520–527.

Gislason, H., and Rice, J.C. 1998. Modelling the response of size and diversity spectra of fish assemblages to changes in exploitation. ICES J Mar Sci, 55: 362–370.

Gislason, H., and Sinclair, M. 2000. Ecosystem effects of fishing. ICES J Mar Sci, 57: 465–791.

Greenstreet, S.P.R. 1996. Estimation of the daily consumption of food by fish in the North Sea in each quarter of the year. Scott Fish Res Rept, 55.

Greenstreet, S.P.R., and Hall, S.J. 1996. Fishing and ground-fish assemblage structure in the north-western North Sea: an analysis of long-term and spatial trends. J Anim. Ecol, 65: 577–598.

Greenstreet, S.P.R., Spence, F.E., and McMillan, J.A. 1999a. Fishing effects in northeast Atlantic shelf seas: patterns in fishing effort, diversity and community structure. V. Changes in structure of the North Sea groundfish assemblage between 1925 and 1996. Fisheries Research, 40: 153–183.

Greenstreet, S.P.R., Spence, F.E., Shanks, A.M. and McMillan, J.A. 1999b. Fishing effects in northeast Atlantic shelf seas: patterns in fishing effort, diversity and community structure. II. Trends in fishing effort in the North Sea by U.K. registered vessels landing in Scotland. Fisheries Research, 40: 107–124.

Hall, S.J. 1999. The effects of fishing on marine ecosystems and communities. Blackwell Science, Oxford.

Hempel, G. (Ed.) 1978. North Sea fish stocks recent changes and their causes. Rapp P-v. Réun Cons int Explor Mer, 172: 1–449.

Hill, L., Mohn, R., Collie, J. S., and Borges, M. F. 2001. Life history characteristics as tools to evaluate changes in exploited fish communities. ICES CM 2001/T:08, 17 pp.

Hill, M.O. 1973. Diversity and evenness: a unifying notation and its consequences. Ecology, 54: 427-432.

Huston, A.H. 1994. Biological Diversity: The Coexistence of Species on Changing Landscapes. Cambridge University Press, 681 pp.

ICES. 2001. Report of the Working Group on the Ecosystem Effects of Fishing Activities. ICES CM 2001/ACME:09. 102 pp.

Jakobsson, J., Astthorsson O.S., Beverton, R.J.H., Bjornsson B., Daan, N., Frank, K.T., Meincke, J., Rothschild, B.J., Sundby, S., and Tilseth, S. 1994. Cod and climate change. ICES Marine Science Symposia, 1–693.

Jennings, S., Reynolds, J.D., and Mills, S.C. 1998. Life history correlates of responses to fisheries exploitation. Proceedings of the Royal Society of London, 265: 1–7.

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

Jennings, S., Alvsvåg, J., Cotter, A.J., Ehrich, S., Greenstreet, S.P.R., Jarre-Teichmann, A., Mergardt, N., Rijnsdorp, A.D., and Smedstad, O. 1999b. Fishing effects in northeast Atlantic shelf seas: patterns in fishing effort, diversity and community structure. III. International fishing effort in the North Sea: an analysis of temporal and spatial trends. Fish Res 40: 125–134.

Jennings, S., Warr, K.J., Greenstreet, S.P.R., and Cotter, A.J.R. 2000. Spatial and temporal patterns in North Sea fishing effort. *In* Effects of Fishing on Non-Target Species and Habitats: Biological, Conservation and Socio-economic Issues. Ed. by M.J. Kaiser and S.J. de Groot, Blackwell Science, Oxford.

Jennings, S., Pinnegar, J.K., Polunin, N.V.C., and Boon, T.W. 2001. Weak cross-species relationships between body size and trophic level belie powerful size-based trophic structuring in fish communities. Journal of animal Ecology, 70: 934–944.

Jennings, S., Pinnegar, J.K., Polunin, N.V.C., and Warr, K.J. 2002. Linking size-based and trophic analyses of benthic community structure. Mar Ecol Prog Ser, 226: 77–85

Jennings, S., Greenstreet, S.P.R., Warr, K.J., and Pinnegar J.K. In prep. Long-term impacts of fishing on the trophic structure of the North Sea fish community: evidence from stable isotope analysis and size-spectra.

Magurran, A.E. 1988. Ecological Diversity and its Measurement. Chapman and Hall, London.

Minawaga M., and Wada, E. 1984. Stepwise enrichment of 15N along food chains: further evidence and the relation between 15N and animal age. Geochemica et Cosmochimica Acta, 48: 1135–1140.

O'Brien, C.M., Fox, C.J., Planque, B., and Casey, J. 2000. Climate variability and North Sea cod. Nature, 404: 142.

Owens, N.J.P. 1987. Natural variations in ^{15}N in the marine environment. Adv Mar Biol, 24: 389–451

Paine, R.T. 1974. Intertidal community structure. Experimental studies on the relationship between a dominant competitor and its principal predator. Oecologia, 15: 93–120.

Pauly, D. and Christensen, V. 1995. Primary production required to sustain global fisheries. Nature, 374: 255–257.

Pauly, D., Christensen, V., Dalsgaard, J., Froese, R., and Torres, F. 1998. Fishing down marine food webs. Science, 279: 860–863

Pauly, D., Palomares, M.A., Froese, R., Sa-a, P., Vakily, M., Preikshot, D., and Wallace, S. 2001. Fishing down Canadian aquatic food webs. Can J Fish Aquat Sci, 58: 51–62

Piet, G. J. 2001. Development of ecological quality objectives for the North Sea Fish community. Working paper WGECO 2001, 32 pp.

Pinnegar, J.K., Jennings, S., O'Brien, C.M., and Polunin, N.V.C. 2002. The effects of exploitation and environmental variability on the trophic structure of the Celtic Sea fish community. J Appl Ecol., in press.

Polunin, N.V.C., and Pinnegar, J.K. in press. Ecology of fishes in marine food-webs. *In* Handbook of Fish and Fisheries. Ed. by P.J. Hart and J.D. Reynolds. Blackwell, Oxford.

Rice, J., and Gislason, H. 1996. Patterns of change in the size spectra of numbers and diversity of the North Sea fish assemblage, as reflected in surveys and models. ICES J Mar Sci, 53: 1214–1225.

Rice, J.C. 2000. Evaluating fishery impacts using metrics of community structure. ICES J Mar Sci, 57: 682–688.

Rogers, S.I., and Ellis, J.R. 2000. Changes in the demersal fish assemblages of British coastal waters during the 20th century. ICES J Mar Sci, 57: 866–881.

Rosenzweig, M.L. 1992. Species diversity gradients: we know more and less than we thought. Journal of Mammalogy, 73, 715–30.

Rosenzweig, M.L. and Abramsky, Z. 1993. How are diversity and productivity related? *In* Species Diversity in Ecological Communities: Historical and Geographical Perspective, pp. 52-65. Ed. by R. Ricklefs and D. Schluter. University of Chicago Press, Chicago.

Rosenzweig, M.L. 1995. Species Diversity in Time and Space. Cambridge University Press, UK. 436 pp.

Smith, T.D. 1994. Scaling fisheries: the science of measuring the effects of fishing, 1855–1955. Cambridge University Press, Cambridge.

Sparholt, H. 1990. An estimate of the total biomass of fish in the North Sea. J Cons int Explor Mer, 46: 200–210

Walker, P.A., and Heessen, H.J.L. 1996. Long-term changes in ray populations in the North Sea. ICES Journal of Marine Science, 53: 1085–1093.

Walker, P.A., and Hislop, J.R.G. 1998. Sensitive skates or resilient rays? Spatial and temporal shifts in ray species composition in the central and north-western North Sea between 1930 and the present day. ICES Journal of Marine Science, 55: 392–402.

Yang, J. 1982. A tentative analysis of the trophic levels of North Sea fish. Mar Ecol Prog Ser, 7: 247–252.

6 EVALUATING THE IMPACT OF FISHING PRACTICES ON NON-TARGET SPECIES

Our Term of Reference (d) requires us to "*in response to the EC DG Fish request for an 'evaluation of the impact of current fishing practices on non-target species, ... and suggestions for appropriate mitigating measures', investigate ways to use data products produced by the Study Group on Discard and By-catch Information for ecosystem management studies [contingent on discard and by-catch from SGDBI being available for further analyses]. Where data are sufficient, evaluate the impact of fishing on non-target species. Identify species and fisheries where mitigative actions may be warranted and, in such cases, propose and justify alternative mitigation measures".*

6.1 Introduction

Although the report of the recent meeting of SGDBI (March 2002) was not yet complete, a CD-ROM with all tabulated information was made available prior to the meeting of WGECO. In scanning this information, WGECO noted that so far SGDBI has concentrated on providing comprehensive discard information for a wide range of fisheries and areas, but has apparently given priority to commercial species that are routinely assessed by ICES. The terms of reference of SGDBI did not specify whether non-target species were included in the task, although such information has been generally collected during discard programmes, and so might be made available in due course. As a consequence of this lack of information on non-target species, it has not been possible for WGECO to make progress with this ToR. However, there are other problems as well because, throughout the ICES area, data on non-target species are, with possibly a few exceptions (e.g., some stocks of rays and skates), structurally insufficient to evaluate impacts of fishing. Moreover, mitigative actions (with the exception of prohibiting particular gears in their entirety) would require specific analyses of spatially disaggregated data for an open-ended list of non-target species that are at least potentially impacted by fisheries. In this section, we therefore describe an approach, with associated data requirements, that might allow progress in the future.

6.2 Evaluating impacts on non-target species

Biomasses of non-target species are difficult to estimate accurately. Without such estimates it is not possible to quantify and evaluate impacts in terms of by-catch mortality. Of course, other information about species' status may be used to at least identify situations where measures to reduce by-catch might be justified. For instance, survey information may indicate prolonged declines over part of their distribution area or even the entire range. In combination with relatively high by-catch rates, particularly if data were spatially disaggregated, it might be possible to isolate cases where fisheries at least contributed to the species' decline. Such information might serve as a basis for advice on by-catch mitigation programmes, particularly if followed up by monitoring the effects of the latter. Nevertheless, evaluation of the impact on individual species might be strengthened considerably, if quantitative discard data could be related to some absolute biomass estimate. However, it would be virtually impossible to collect detailed information on population structure for most of these species—whether partly landed or totally discarded—that would allow analytical assessments.

Despite many problems and shortcomings, the methods of Yang (1992) and Sparholt (1990) to transform qualitative survey data into absolute biomass estimates of all species recorded are considered to provide the best descriptions of the North Sea fish community obtained so far. Given that the data base has been extended enormously since the 1980s, an update seems urgently required. However, an additional problem is that in fact a wide variety of surveys have been carried out with different gears with varying catchabilities for every single species. Consequently, different data sets may provide different relative biomasses and this obviously reduces their usefulness for management purposes, because their representativeness can always be argued. What seems needed, therefore, is a coherent analysis of all surveys combined by developing suitable raising factors for comparing catches per swept area taken by different gears. This in itself is a major exercise, that has to be repeated for each major management area.

Non-target species lists are essentially open-ended, because any vagrant species may accidentally end up in a fishing net. Many of these incidental by-catch species probably should not concern local management, because their stocks depend predominantly on factors beyond local control. Similarly, the species list may be suitably truncated by excluding species that are too small to be caught effectively or that are typical of unfishable areas such as rocky coasts. By defining an appropriate resident and potentially impacted fish community, the total amount of work may be reduced considerably.

Once biomass (B) estimates of standing stocks are available (preferably on an annual basis to reveal trends), quantitative information on (discarded) by-catches (C) may be used to calculate C/B ratios. Their ranking may not be entirely representative of the true impact on each species, because impact is related to the catch over production (P) ratio rather than catch over standing stock and P/B ratios vary with maximum age and size. Nevertheless, the C/B ratio would provide an objective first estimate of the impact of the fishery, preferably by fleet. Comparison with similar estimates for regularly assessed commercial species might further help to identify the significance of the estimated impact ratios.

6.3 Mitigation measures

If the estimated impact ratio for a particular non-target species leads to management concern, the usual mitigation measures (reduction of fishing effort, gear restrictions, closed areas) might be applied. However, the most promising measures are probably those that interfere least with existing fishing practices, such as closing seasons when, and areas where, the by-catch problem is largest and commercial yield is smallest. To evaluate appropriate mitigation measures, it is therefore important to have by-catch data available at disaggregated temporal and spatial scales.

6.4 Discussion

WGECO first reviewed information on discards in 1994 (ICES, 1994), when amounts discarded in the North Sea were assessed and information on consumers of the discards was reviewed. WGECO noted that while there had been a number of discard studies, recording methods had varied considerably and that data were usually only available in aggregated format. These formats were often incompatible and it was therefore not possible to generate a wider picture of discarding. ICES (1994) found that paradoxically more data were available on the amount and pattern of discarding of non-commercial species than were available on commercial species. A strong recommendation was made that disaggregated data should be made available from the discard studies and these should be compiled internationally to gain the wider view needed of the impact of fishing activities. This would help both fisheries advice and studies of ecosystem effects. The group briefly revisited the issue in 1996 and found that there had been little progress (ICES, 1996).

In 1997, OSPAR requested advice from ICES on quantities of discards by gear type and OSPAR regions for commercially exploited stocks of fish and shellfish. ICES tasked WGECO with compiling this information on the basis of information to be supplied by relevant fish stock assessment groups. WGECO made a number of recommendations, noting some progress since 1994 in starting studies of discards, but raising concerns that information from these studies was not being made available to ICES for use in advice on fish stocks and the ecosystem (ICES, 1998). Two points became clear with regard to how ICES deals with data on discards. WGECO reported “First of all, information ICES has and uses on discards is so incomplete that it compromises the quality of some of the analytical work,… Second, there are diverse perspectives on discard issues, and the diverse perspectives need to be bought more fully into the discussions ICES has on discards” (ICES, 1997). WGECO therefore recommended that a Study Group be formed to address discard issues with membership drawn from both assessment working groups and the wider environmental community. At least partially as a result of this, SGDBI was established.

SGDBI has addressed the analytical issues raised by WGECO in ICES (1998), but it is unfortunate that its membership and terms of reference have been solely orientated towards these issues. The wider environmental community and its needs have not been included in the work of SGDBI. Consequently, eight years after WGECO first reviewed the issue of discards and six years after ICES was first asked for information on by-catch and discards of non-target species (and their wider effects, such as on food for other biota), it is regrettable that these questions can still not be fully considered. WGECO has not even been able to assess whether the data collected on discards suit the purpose of evaluating effects of fishing on non-target species. It is worth noting that this time the request came from fisheries managers concerned with the environmental effects of fishing, and additionally asking for advice on mitigative measures.

WGECO still thinks it important that ICES be able to provide advice in this area and therefore recommends that further efforts be made to integrate the work on discards and by-catch between the fish stock assessment and environmental sides of ICES. Estimates of partial by-catch mortality rates are the strongest basis for advice on management of non-target species, yet a very large amount of work is required before ICES can have the data and tools ready for such management advice. The integration of different sets of survey data into biomass estimates for the individual species composing regional fish communities by developing appropriate raising factors requires sophisticated statistics. This task might be given to a Study Group to be established in 2003 for this specific purpose. The group could also address the options for estimating trends in abundance of non-target species, particularly on regional scales, for use in cases where the more difficult task to estimate absolute abundances could not be provided in the short term. If SGDBI could take on a spatially disaggregated tabulation of discard information on non-target species as its first priority in 2003, WGECO could have a first go at evaluating actual impacts and potential mitigation measures in 2004.

Recommendations

To further evaluate the impacts of fisheries on non-target species and potential mitigation measures, WGECO recommends that:

- SGDBI processes, as a priority, the discard data on non-target species, disaggregated by ICES rectangle and month;
- A new Study Group be established and tasked with the development of the methodology to integrate information on the North Sea fish community derived from different surveys and to derive time series of absolute biomass, with associated estimates of uncertainty, from the integrated set for as many non-target species as possible;
- Once the above are complete, that WGECO be asked to review the impacts of fisheries on non-target species and potential mitigation measures.

Justification:

the information to be collected is essential if ICES is to provide advice on fisheries impacts on a wide variety of non-target species. It is suggested that the SG concentrates on the North Sea first, because a wide variety of surveys have been maintained over many years, using different gears that all provide different ‘pictures’ of the actual community present. The methods developed should be applied subsequently to surveys in other areas.

References

ICES. 1994. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 1994/Assess/Env:1.

ICES. 1996. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 1996/Assess/Env:1.

ICES. 1998. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 1998/ACFM/ACME:1.

Sparholt, H. 1990. An estimate of the total biomass of fish in the North Sea. Journal de la Conseil international de l'Exploration de la Mer, 46: 200–210.

Yang, J. 1992. An estimate of the fish biomass in the North Sea. J Cons int Explor Mer, 40: 161–172.

7 THE DISTRIBUTION OF COLD-WATER CORAL AND THREATS FROM HUMAN ACTIVITIES

7.1 Information on the distribution of cold-water coral in the ICES area

WGECO reviewed the March 2002 version of the Report of the Study Group on Mapping the Occurrence of Cold- Water Corals (SGCOR) (ICES, 2002). *Lophelia* appears to prefer waters with a temperature of between 4 ^{o}C and 12 ^{o}C, with relatively high water flow. *Lophelia* appears to grow in areas raised above the sedimentary sea floor, on rock, boulders, sand mounds or man-made structures. These conditions occur widely in the Northeast Atlantic including on the continental shelf in a number of more northerly areas. Broadly though, these conditions occur at shallower depths in some Norwegian fjords (within 40 m of the surface in Trondheimsfjorden (Strømgren, 1971; Rapp and Sneli, 1999)) and at much greater depths off the Iberian peninsula. ICES (2002) provides a good description of current knowledge of the distribution of *Lophelia pertusa* in the ICES area and should be referred to for further details. The distribution of *Lophelia* in the ICES area is summarised in Figures 7.1.2–7.1.8 that have been taken direct from the ICES (2002). There are a number of projects looking at various aspects of cold-water corals presently under way in ICES waters (see Section 7.2, below), and an update was provided to the meeting as a working paper from Iceland (Steingrimsson, 2002), an as a poster (Unnithan, 2001). WGECO noted that it would be useful to update the cold-water coral report with new information immediately prior to the June 2002 ACE meeting.

Figure 7.1.1. Fragments and larger pieces of dead *Lophelia pertusa* near Iverryggen on the Norwegian continental shelf at 190 m depth. Photo taken from a height of about 2 m above the seabed on 17 May 1999. The bottom substrate is severely disturbed and the trench running across the picture from centre left to top right is apparently caused by towed trawl gear. From Fosså *et al.* (in press).

Figure 7.1.2. Location map for detailed maps of the distribution of *Lophelia* in European waters (Figures 7.1.3-7.1.8), based on information compiled by Freiwald (1998), and Hovland and Mortensen (1999).

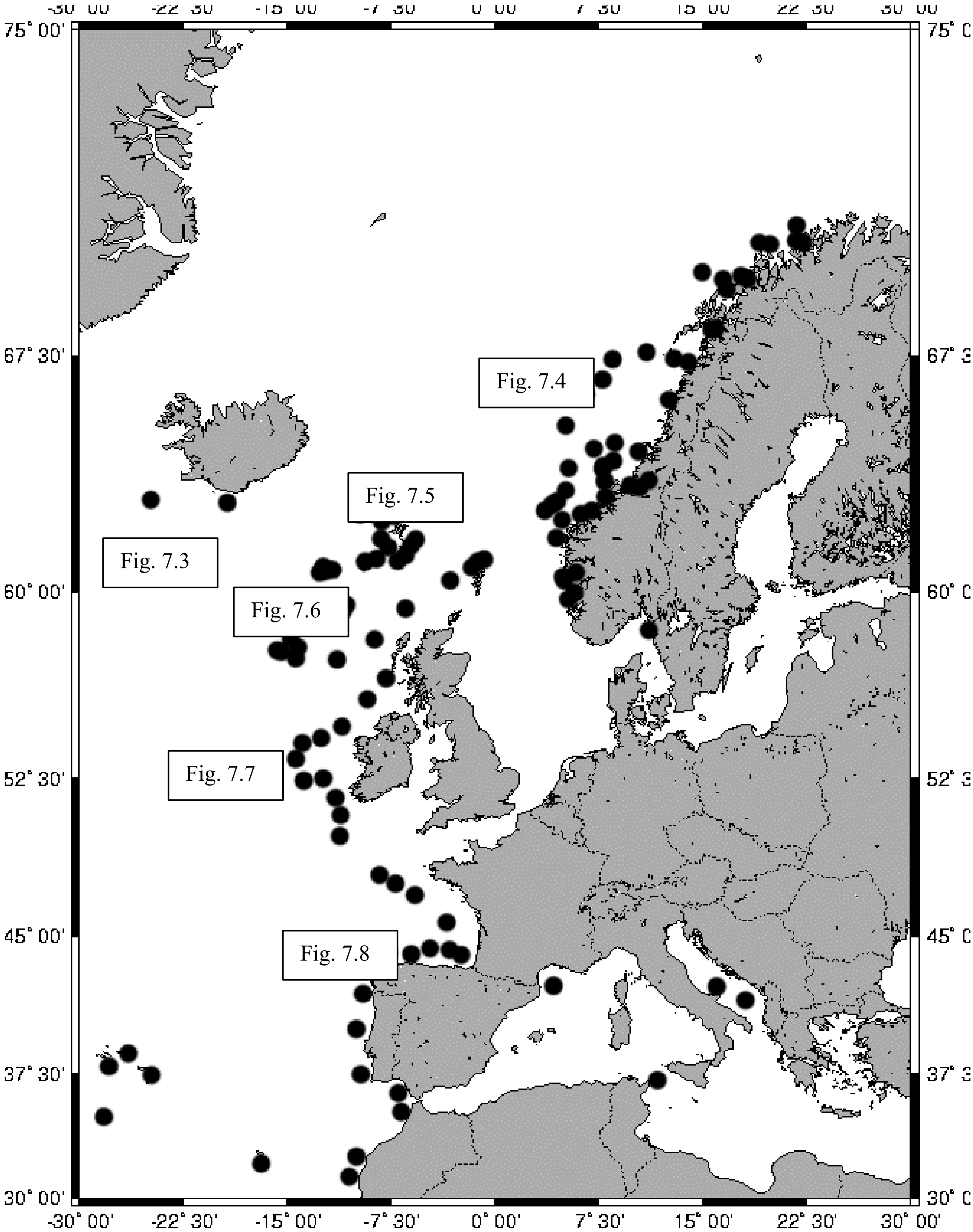


Figure 7.1.3. Distribution of records of *Lophelia pertusa* made during the BIOICE programme in Icelandic waters (map provided by S.A. Steingrímsson).

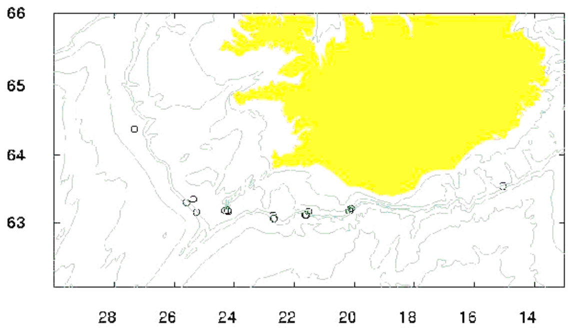


Figure 7.1.4. Distribution of *Lophelia pertusa* (dots) and major trawl grounds (blue) in Norwegian waters, showing the degree of overlap between coral and trawling distribution. Two areas on the shelf, Sula and Iverryggen (red), are protected from trawling gear. Map from J.H. Fosså, Institute of Marine Research, Bergen.

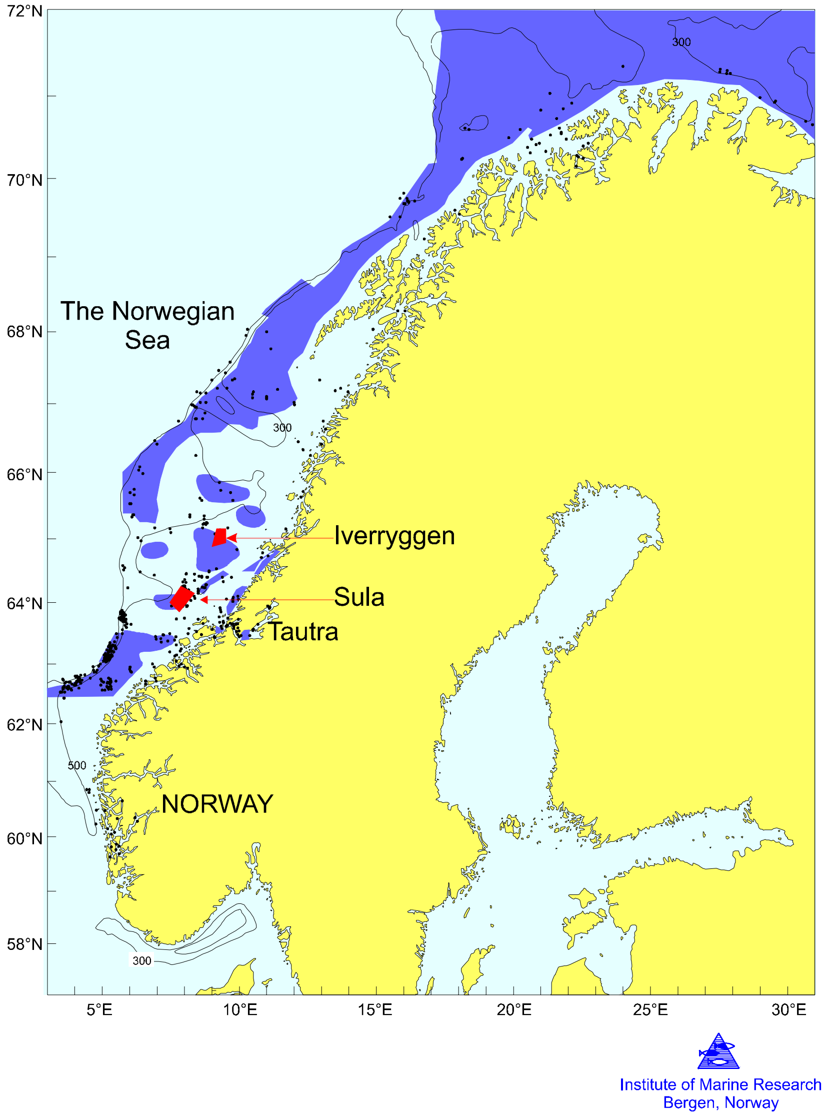


Figure 7.1.5. Distribution of current (solid green) and past (hatched green) areas containing *Lophelia pertusa* reefs in waters around the Faroe Islands (S. H. í Jákupsstovu). It is assumed that reefs in the hatched areas have been lost through fishing activity. The red lines are areas presently closed for trawling, for fisheries management reasons.

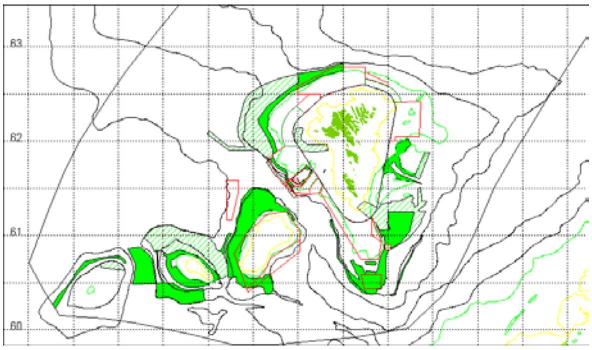


Figure 7.1.6. Distribution of *Lophelia pertusa* (dots) on the Faroes Bank (Magnussen, in press).

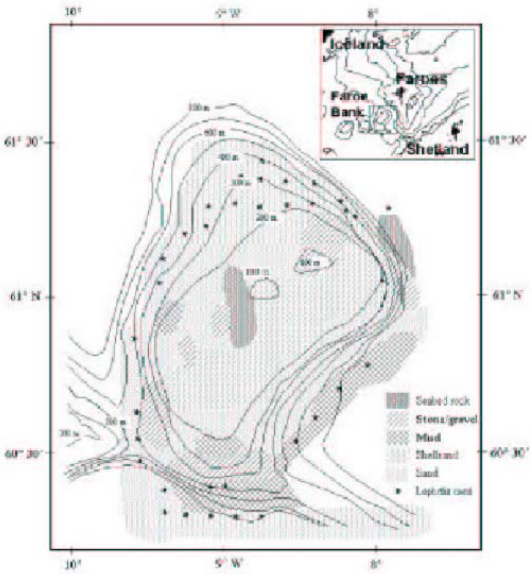


Figure 7.1.7. Potential and actual distribution of *Lophelia pertusa* in northwestern waters of the United Kingdom (map courtesy of Southampton Oceanography Centre).

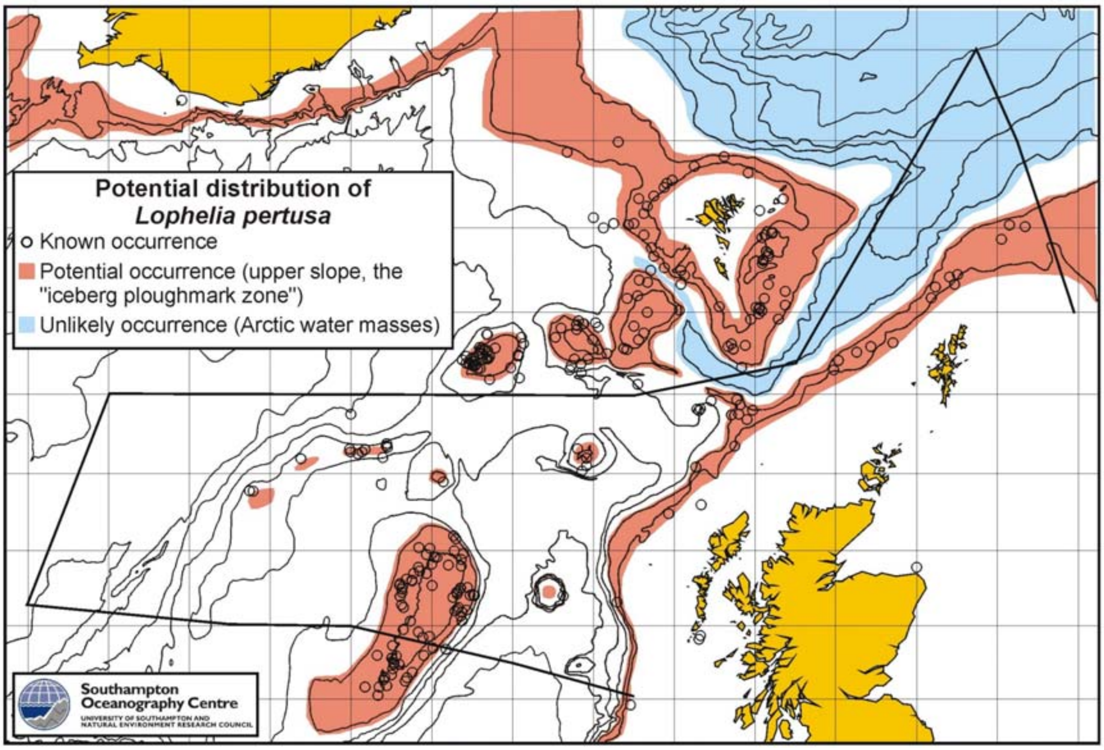
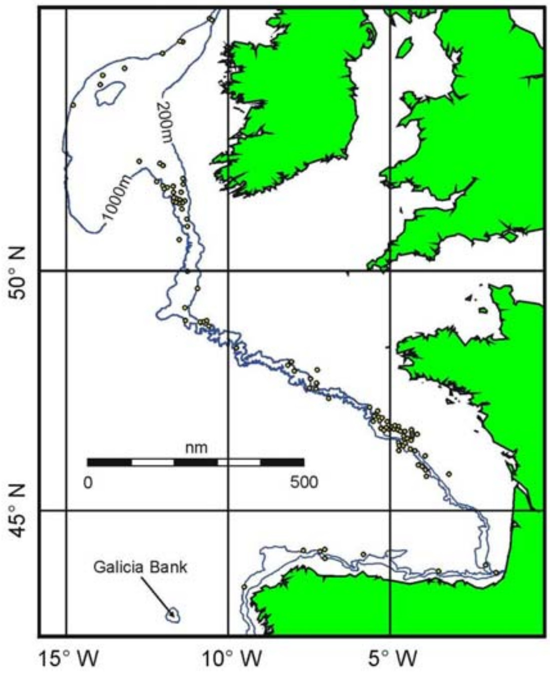


Figure 7.1.8. Early observations of the distribution of deep-water corals (mainly *Lophelia pertusa*) in the Porcupine Seabight and Bay of Biscay areas, compiled from the records of commercial fishing vessels (adapted from Teichert, 1958; after Joubin, 1922a, b). The Galicia Bank is also indicated; this site is now known to hold a significant population of *Lophelia pertusa* (G. Duineveld, pers. comm.).



7.2 Current projects on cold-water corals

Several projects are under way examining aspects of *Lophelia* or carbonate mound biology or geology. There are three EU-funded projects: the Atlantic Coral Ecosystem Survey (ACES), now entering its last year; environmental controls on mound formation along the European Margin (ECOMOUND) and its geological equivalent GEOMOUND. ACES study sites are: the Galicia Bank, the Porcupine slope, Rockall Trough, Kosterfjord (Sweden) and the Sula Ridge (Norway).

Information on the location of deep-water corals is being gathered as part of a joint multibeam survey of bottom types around the Irish coast carried out by the Geological Survey of Ireland/Marine Institute and the Irish Sea Fisheries Board (BIM). A monitoring programme for coral by-catch has also been put in place off Ireland.

Off Icelandic coasts, the Marine Research Institute is mapping the historic and present distribution of coral grounds in order to evaluate the extent of possible degradation of *Lophelia* reefs. Retired fishermen who were actively fishing around 1950-1960 will be interviewed in order to add to already published records. Fishermen currently working will also be interviewed to establish the present distribution of *Lophelia* grounds.

7.3 Impacts on cold-water corals

7.3.1 Trawling

Trawling is widespread in areas holding *Lophelia*. Photographic and acoustic surveys have recently located trawl marks at 200–1400 m depth all along the Northeast Atlantic shelf break area from Ireland, Scotland and Norway (Rogers, 1999; Fosså *et al.,* 2000; Roberts *et al.,* 2000; Bett, 2000).

There have been a number of documented instances of damage to *Lophelia* reefs in Northwest European waters. These, though, must represent a small proportion of the number of instances when such reefs have been damaged, given how widespread trawling has been, and the amount of habitat (Section 7.1) that is potentially suitable for corals in the Northeast Atlantic. Another indication that damage to corals by trawling has been widespread is that many records of occurrence come from commercial trawlers hauling up broken pieces of coral.

The most obvious impact of trawling is mechanical damage caused by the gear itself. The impact of trawled gear kills the polyps and breaks up the reef structure. The breakdown of this structure will alter the hydrodynamic and sedimentary processes and recovery may not be possible, or could be seriously impaired. It may also cause loss of shelter around the reef and organisms dependent on these features will have a less suitable habitat. The scale of effects depends on the scale and frequency of trawling operations. Damage may range from a decrease in the reef size, and a consequent decrease in abundance and diversity of associated fauna, to a complete disintegration of the reef and its replacement with a disturbed low-diversity community (Fosså *et al.,* 2000). Trawling may also have the effect of evening out the seabed by scraping off high points and infilling lows, as well as redistributing boulders. Since *Lophelia* requires some of the high points to grow initially, the seabed habitat following trawling may become unsuitable.

Trawls also cause resuspension of sediments that could affect corals growing downstream (including entrapment in the coral framework). Such impacts may be proportionately greater in high-relief mound areas such as in the Porcupine Seabight, where trawling over the mounds is uncommon owing to the risk of gear damage and large unwanted by-catch. However, the sediment areas immediately adjacent to the mounds are heavily trawled. Quantitative evidence for such impacts was found by the French ROV Victor during the "Caracole" cruise in August 2001 (A. Freiwald, pers. comm.).

Fosså *et al.* (2000) estimated that between a third and a half of Norway's *Lophelia* reefs are damaged or affected by fishing. Damage is illustrated from a number of areas by comparing photographs (damage is difficult to quantify by sampling because sampling itself also causes damage). Fosså *et al.* (in press) describe these surveys. To distinguish natural decay from impacts by human activities, such as bottom trawling, they looked for broken living colonies tilted, turned upside down and/or in unexpected/awkward positions on levelled sea bottom. The remains of trawl nets among corals and recent furrows or scars in the sea bottom were also taken to be evidence of trawling activity.

Three localities on Storegga (continental shelf break between 62° 30' N and 63° 50' N) were inspected between 1998 and 1999: Aktivneset, Korallneset and Sørmannsneset. During 1999, two localities were inspected on the shelf: Maurdjupet and Iverryggen. All these localities and surrounding areas are subject to extensive bottom trawling.

Two inspections with ROV were made at Sørmannsneset, covering a vertical range from 370 m to 225 m and distances between 2.5 km and 2.9 km. The observations confirmed that the most severe damage occurred at the shallowest depths (200 m) as crushed remains of *Lophelia* skeletons were spread over the area while living corals were rarely found. Many signs of trawling were found including wires and remains of a trawl net entangled with corals. In addition, sonargrams from the side-scan sonar detected furrows penetrating into areas of damaged corals. These were interpreted as furrows as caused by trawl doors or other parts of a trawl gear cutting through the surface of the bottom. At Korallneset, almost 2.6 km of the sea bottom was inspected between 305 m and 205 m depth. Almost all corals observed were crushed or dead. Aktivneset is subject to heavy trawling and the ROV-inspection showed this location to be very rich in corals all along a 7 km ROV-transect between 350 m and 270 m depth. The reefs were neither large nor high, but smaller colonies were spread over large areas. However, damage was evident and furrows in the seabed were observed. Damage at Maurdjupet was severe, especially on the slopes of a smaller basin (or depression) in the shelf. Five inspections at Iverryggen revealed severe damage to colonies of *Lophelia* and other corals such as gorgonians (Figure 7.1.1). Every inspection verified damage exhibiting all stages of degradation, e.g., from almost intact living coral colonies to completely crushed reefs.

The Darwin Mounds were discovered using remote sensing techniques in May 1998 during surveys funded by the oil industry and steered by the Atlantic Frontier Environment Network (AFEN), a UK industry-government group (Masson and Jacobs, 1998). They have been further investigated in June 1998 (Bett, 1999), August 1999 (Bett and Jacobs, 2000) and twice during summer 2000 (Bett *et al.,* 2001; B. Bett, pers. comm.). Instruments deployed during the studies have included side-scan sonar, stills and video cameras, and piston corers.

The Darwin Mounds are vulnerable to damage from bottom trawling, and evidence of new damage was visible over about half of the Darwin Mounds East during summer 2000 (Wheeler *et al.* 2001). This damage was visible as smashed coral strewn on the seabed along with visible parallel scar marks. Given that *Lophelia pertusa* appears to need (or favour) the elevation provided by sand mounds for growth in this area, it seems likely that this damage will be permanent. This site must be regarded as at particularly high risk of further permanent damage.

Hall-Spencer *et al.* (2002) found significant coral by-catch in five out of 229 hauls observed of French trawlers working in the Porcupine Seabight area. Trawling in this area is undertaken by French, Irish and Scottish vessels for mixed species such as orange roughy, roundnose grenadier, blue ling, black scabbard and sharks. Trawling for orange roughy has been shown to have caused major destruction of seamount corals in Tasmania and New Zealand (Koslow *et al.*, 2001).

7.3.2 Demersal longlining

Although lost longlines have been observed on video surveys of coral areas, no evidence of actual damage to reefs has been shown. It seems likely that some coral branches could be broken off by longlines. In Icelandic waters, longline vessels seek out coral reefs in search of species using the structures as habitat (Steingrimsson 2002). Species thus targeted include tusk *Brosme brosme*, ling *Molva molva*, blue ling *Molva dypterygia* and various species of redfish *Sebastes* spp. Off Ireland longlining is undertaken by Norwegian vessels for ling and tusk. It is also undertaken by Spanish, UK and Irish vessels for hake, sharks, ling and forkbeards, but few data are available.

7.3.3 Gill and tangle netting

The surveys referred to in Section 7.3.1 have also found evidence of damage from gill and tangle netting. The video inspections of the Storegga, Norway found lost (and ghostfishing) gillnet, an anchor and a buoy. The nets and anchor-ropes may sometimes break down and tilt parts of the colonies. Video surveys by Southampton Oceanography Centre in 1999 and by IFREMER in 2001 showed gillnets ghost fishing on carbonate mounds/*Lophelia* reefs on the western edge of the Porcupine Bank in ICES Division VIIc. The Spanish have a traditional gillnet hake fishery in an area 60 miles SW of Valencia in a coral-rich area.

7.3.4 Sediment input from drilling operations

Corals are known globally to be sensitive to increased levels of sedimentation. Fine sediments can clog the polyps, leading to reduced feeding rates and, in some cases, permanent damage (Reigl, 1995). *Lophelia* appears to grow in areas raised above the sedimentary sea floor, on rock, boulders, sand mounds or man-made structures. This habit is probably to avoid clogging of the polyps. Concern has been expressed about discharges of drill cuttings from hydrocarbon exploration in deep water (Rogers, 1999). A research project to assess the effects of sand deposition on *Lophelia* behaviour has been undertaken by the Scottish Association for Marine Science. Early results indicate that polyp behaviour is naturally variable, but that sand deposition at 0.1 mg cm^{-2} min^{-1} significantly reduced the level of polyp expansion when four sediment-exposed polyps were compared with six control polyps (J.M. Roberts, pers. comm.). Discharged drill cuttings could therefore potentially affect *Lophelia*. However, the amount of cuttings locally discharged from hydrocarbon drilling to the sea may be reduced to a low level by shipping the cuttings to the shore; it would thus be possible to drill near to *Lophelia* reefs without placing the reefs at risk.

7.3.5 Chemical input

Similar concerns as to drill cuttings apply to chemical discharges from the offshore industry. No research is known of the effects of chemicals on *Lophelia* but nevertheless, as with cuttings, it would be entirely possible to conduct offshore hydrocarbon operations without discharge of chemicals. In the depths where *Lophelia* reefs are commonly found, accidental discharges on the surface of the sea are likely to be heavily diluted by the time any chemical might reach the seabed.

7.3.6 Summary

Trawling-induced damage to deep-water coral reefs has been proved in several areas, with perhaps the worst damage being evident presently on the reefs in shallower waters off Norway. However, there are several older records from continental shelf seas that appear to have suitable hydrographic conditions for *Lophelia*, and it seems likely that persistent trawling in these waters has extirpated it. This suggestion is supported by recent observations of *Lophelia* growing on (undisturbed parts of) oil platforms (Bell and Smith, 1999). Deeper reefs off Ireland and southwards seem not to have suffered the same scale of damage, but are nevertheless vulnerable. The effects of other human activities are likely to be minor in comparison to those of trawling.

7.4 Mitigation/protection of corals from human activities

The only way to completely prevent damage to areas of deep-water coral is to accurately map these areas and then close them to fisheries. Such closure may have other benefits, as a letter from the Scottish Fishermen's Association in IntraFish recently expressed the need for better coral distribution maps so that fishermen can avoid these areas and thereby the high costs associated with damaged nets and poor fish quality. In Sweden, two reef areas in the Kosterfjord are now protected and management measures have been agreed with local fishermen who now avoid fishing in the area.

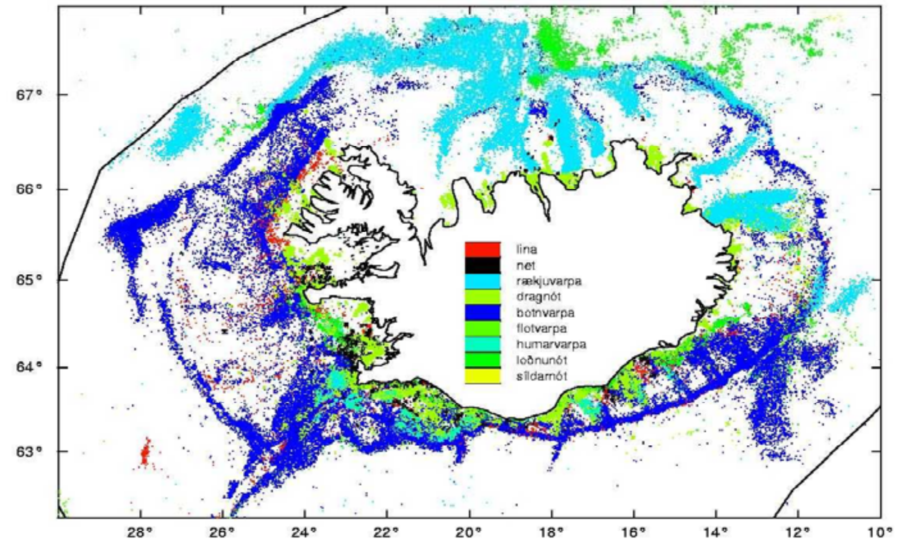
In addition, in EU waters, the Habitats Directive requires the conservation of reef habitat. This is widely interpreted to include *Lophelia* coral reefs. No EU Member States yet have the legal powers to designate Special Areas of Conservation (SAC) beyond territorial waters (12 n.m.), but some, including the UK and Ireland, are expected to have such powers within the next year. The understanding in the UK is that once a candidate SAC has been notified to the European Commission, the Commission will be duty bound to protect that SAC from harm from those activities which it has exclusive competence to regulate (e.g. fisheries). The relevant Minister (Margaret Beckett, Secretary of State, Department for Environment, Food and Rural Affairs) in the UK has indicated (23 October 2001) that the Darwin Mounds will be in the vanguard of any list of candidate sites notified to the European Commission.

7.4.1 Closed areas to trawling

The Icelandic study on the location of coral reefs and of trawl and longline fisheries (Steingrimsson, 2002) perhaps gives a good example of a way forward to determining suitable areas to close to fisheries if protection of *Lophelia* is required.

An area off the south and west coasts of Iceland was defined enclosing the known distribution limits of *Lophelia* in Icelandic waters ("coral" area). Fishing effort data for 1999 and 2001 for otter trawling and longlining occurring within the area were obtained from the effort database (Figure 7.4.1.1). Gear type, position (latitude, longitude) and catch composition (species, catch (kg)) were obtained from each haul. It is known that otter trawls avoid coral areas while longliners seek them out.

Figure 7.4.1.1. Distribution of fishing effort by all gears in Icelandic waters in 1997 (from Steingrimsson, 2002). Lina = longline, Net = Gillnet, Rækjuvarpa = Prawn trawl, Dragnot = Seine net, Botnvarpa = Otter trawl, Flotvarpa = Pelagic trawl, Humarvarpa = Nephrops trawl, Lodnunot = Capelin nets, Sildarnot = Herring nets.



For each rectangle of 1' latitude and 1' longitude, the degree of overlap (O) between fleets (otter trawlers and long-liners) was estimated using the following equation (see Horn, 1966)

$$O = 2\left(p_{aj}\,p_{bj}\right)/\left({p^2}_{aj} + {p^2}_{bj}\right)$$

where P_{aj} = the proportion of haul positions in square j of fleet a. The coefficient ranges between 0 and 1; a value of 0 indicates that both fleets are fishing in completely different squares and consequently a value of 1 indicates that the effort of both fleets was exactly identical in a given square. Squares with an overlap coefficient close to 0 were identified and the area around them defined as possible *Lophelia* grounds.

The defined areas were examined further for spatial distribution of fishing effort (proportion of effort within the defined "coral" area) of both fleets (otter trawlers and longliners) during 2001. Five areas were identified where no overlap occurred between the two fleets both in 1999 and 2001 (Figure 7.4.1.2). The small-scale distribution of fishing effort within the five areas showed that insignificant otter trawling took place in 2001. However, the effort of longliners was relatively much higher. Detailed examination reveals that there are some areas where only longlining occurs. Where no overlap between fleets occurred and only longline was used, species composition of the catch and their relative abundance (% total catch) was estimated. These areas have catches characterised by *Lophelia*-associated species and are thus likely to have concentrations of *Lophelia* reefs. These areas are also likely to encounter less resistance from trawl fishermen if they are declared closed to trawl netting.

Detailed information such as that shown for Iceland is not available for EU waters. Neither log-book nor satellite-derived data have been released beyond national authorities. Without such information, the identification of areas likely to cause least resistance from fishermen to closure will be impossible.

Figure 7.4.1.2. Five areas where there was low overlap between otter trawls and longlines in the "coral area" to the south of Iceland. Hauls in 1999 (+) and 2001(□). From Steingrimsson (2002).

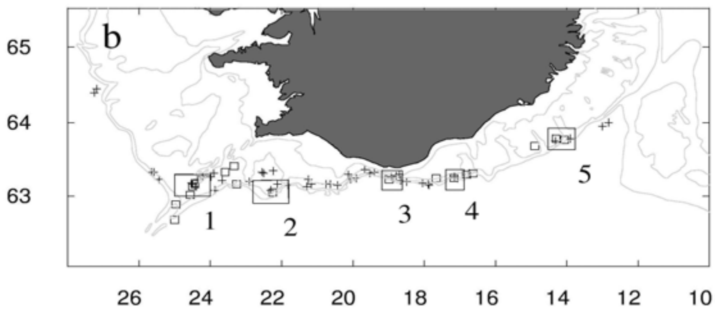
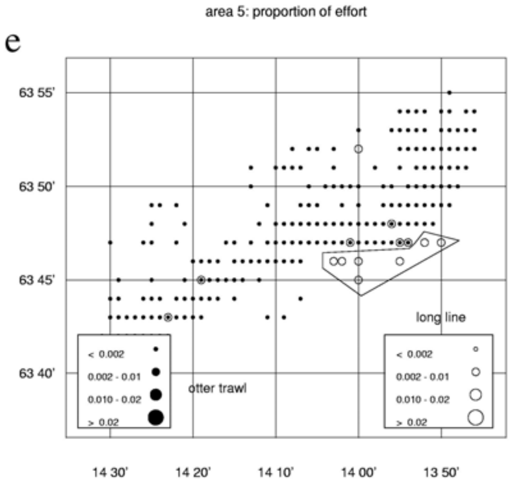
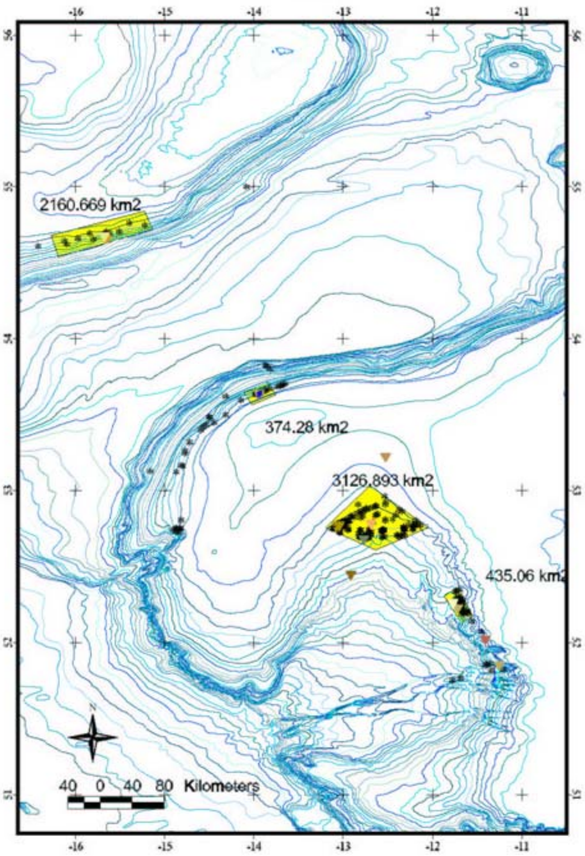


Figure 7.4.1.3. Detail of area 5 (Figure 7.4.1.2) showing an area where only longline fishing occurs. From Steingrimsson (2002).



Irish Deep Water Coral Reef Task Force (2002) has identified four areas suitable for closure to trawl fisheries (Figure 7.4.1.4).

Figure 7.4.1.4. Areas to the west of Ireland containing the best examples of carbonate mounds and *Lophelia* reefs, and suitable for closure to trawl fisheries (Irish Deep Water Coral Reef Task Force, 2002).



7.5 Recommendations

WGECO understands that ACE will be providing further advice on this issue to the European Commission following its June 2002 meeting; it also seems likely that this issue will be visited again in the future. WGECO makes the following recommendations to ensure that both the near- and long-term advice on *Lophelia* is the best possible:

1) As several studies of *Lophelia* are under way, new information is becoming available at relatively frequent intervals. SGCOR should be asked to update their report on cold-water coral distribution for the June 2002 ACE meeting.
2) In order to best tailor advice to actual fishing pressure, ICES should endeavour to acquire access to detailed, suitably depersonalised, data on the location of fishing effort in areas known or likely to contain *Lophelia*.
3) In order to add knowledge on distribution and trawling impact, by-catch recording schemes should include records of *Lophelia*.
4) ICES should continue to advise on the most appropriate areas to close to trawl fishing.

7.6 References

Bell, N., and Smith, J. 1999. Coral growing on North Sea oil rigs. Nature, 402: 601.

Bett, B.J. 1999. RRS Charles Darwin cruise 112C Leg 2, 19 May–24 June 1998. Atlantic Margin Environmental Survey: seabed survey of deep-water areas (17th round tranches) to the north and west of Scotland. Southampton Oceanography Centre, Cruise Report No 25, 171 pp.

Bett, B.J. 2000. Signs and symptoms of deep-water trawling on the Atlantic Margin. Man-Made Objects on the Seafloor. *In* Man-Made Objects on the Seafloor 2000, pp. 107–118. The Society for Underwater Technology, London. (ISBN 0 906940 36 0).

Bett, B.J., and Jacobs, C.L. 2000. RRS Charles Darwin cruise 119C leg B, 13 August–24 September, 1999. White Zone (WhiZ) Environmental survey: Seabed survey of the deep waters to the north and west of Shetland. Southampton Oceanography Centre Cruise report. Report to the UK Department of Trade and Industry.

Bett, B.J., Billett, D.S.M., Masson, D.G., and Tyler, P.A. 2001. RRS Discovery cruise 244, 07 Jul–10 Aug 2000. A multidisciplinary study of the environment and ecology of deep-water coral ecosystems and associated seabed facies and features (The Darwin Mounds, Porcupine Bank and Porcupine Seabight). Southampton Oceanography Centre, Cruise Report No. 36, 108 pp.

Fosså, J.H., Furevik, D.M., Mortensen, P.B., and Hovland, M. 1999. Effects of bottom trawling on *Lophelia* deep water coral reefs on the continental shelf of Norway. Poster and abstract in ICES/SCOR Symposium on Ecosystem Effects of Fishing. Montpellier, France, 15–19 March 1999.

Fosså, J.H., Mortensen, P.B., and Furevik, D.M. 2000. *Lophelia*-korallrev langs Norskekysten forekomst og tilstand. Fisken og Havet 2–2000. Havforskningsinstituttet, Bergen.

Fosså, J.H., Mortensen, P.B. and Furevik, D.M. in press. The deep-water coral *Lophelia pertusa* in Norwegian waters: distribution and fishery impacts. Hydrobiologia.

Freiwald, A. 1998. Geobiology of *Lophelia pertusa* (Scleractinia) reefs in the north Atlantic. Habilitation thesis, University of Bremen. 116 pp.

Hall-Spencer, J., Allain, V., and Fosså, J.H. 2002. Trawling damage to NE Atlantic ancient coral reefs. Proceedings of the Royal Society of London, B. Online paper 01PB0637.

Horn, H.S. 1966. Measurement of “overlap” in comparative ecological studies. American Naturalist, 100: 419–424.

Hovland, M., and Mortensen, P.B. 1999. Norske korallrev og prosesser i havbunnen. Jon Grieg forlag. 155 pp.

ICES. 2002. Report of the Study Group on Mapping the Occurrence of Cold-Water Corals (March 2002). ICES CM 2002/ACE:05.

Irish Deep Water Coral Reef Task Force. 2002. The CFP Review – the case for deep water coral conservation. Presentation (copy available to WGECO).

Joubin, L. 1922. Distribution géographique de quelques coraux abyssaux dans les mers occidentales européennes. Compte rendu hebdomadaire des séances de l'Académie des sciences, 175: 930–933.

Koslow, J.A., Gowlett-Holmes, K., Lowry, J.K., O'Hara, T., Poore, G.C.B., and Williams, A. 2001. Seamount benthic macrofauna off southern Tasmania: community structure and impacts of trawling. Marine Ecology Progress Series, 213: 111–125.

Magnussen, E. in press. The demersal fish assemblages of the Faroe Bank: species composition, distribution, biomass spectrum and species diversity. Marine Ecology Progress Series.

Masson, D.G., and Jacobs, C.L. 1998. RV “Colonel Templar” cruises 01 and 02/98, 22 Apr–18 May, 20 May–18 Jun 1988. TOBI surveys of the continental slope north and west of Scotland. Southampton Oceanography Centre Cruise Report. AFEN UKCS 17th Round Atlantic Margin Environmental Survey Data CD-ROM.

Rapp, H.T., and Sneli, J.A. 1999. *Lophelia pertusa* - myths and reality (abstract only). 2nd Nordic Marine Sciences Meeting, Hirtshals, Denmark, 2–4 March 1999.

Reigl, B. 1995. Effects of sand deposition on sclerectinian and alcyonarian corals. Marine Biology, 121: 517–526.

Roberts, J.M., Harvey, S.M., Lamont, P.A., and Gage, J.A. 2000. Seabed photography, environmental assessment and evidence for deep-water trawling on the continental margin west of the Hebrides. Hydrobiologia, 44: 173–183.

Rogers, A.D. 1999. The biology of *Lophelia pertusa* (Linnaeus 1758) and other deep-water reef-forming corals and impacts from human activities. International Review of Hydrobiology, 84: 315–406.

Steingrimsson, S.A. 2002. Potential coral reefs off the south coast of Iceland. Working paper to ICES Working Group on Ecosystem Effects of Fishing Activities meeting 2002.

Strømgren, T. 1971. Vertical and horizontal distribution of *Lophelia pertusa* (Linne) in Trondheimsfjorden on the west coast of Norway. Det Kongelige Norske Videnskabers Selskabs Skrifter, 6: 1–9.

Teichert, C. 1958. Cold- and deep-water coral banks. Bulletin of the American Association of Petroleum Geologists, 42: 1064–1082.

Unnithan, V. 2001. Caracole cruise August 2001 Porcupine Seabight and Rockall trough cruise overview. Unpublished poster.

Wheeler, A.J., Billett, D.S.M., Masson, D.G., and Grehan, A. 2001. The impact of benthic trawling on NE Atlantic coral ecosystems with particular reference to the northern Rockall Trough. ICES CM 2001/R:11.

8 COMPARING THE STRUCTURE OF ECOSYSTEMS

This section addresses our ToR (f) to "consider the report of the former Planning Group on Comparing Structure of Marine Ecosystems in ICES Area and specifically advise on the areas to be used in ecosystem comparisons and the meta-data available for comparisons".

The report of the Planning Group provides a wide range of interesting ideas, but the conclusions reached underline the impediment to progress:

- We don't know how to (generically) compare ecosystems.
- But if we want to compare aspects, we should aim for linking these aspects in some coherent way.

Ecosystems are essentially a construct of the human mind to split up a diverse and continuous environment into tangible discontinuous units. The only strictly trustworthy scientific use of the ecosystem concept is when the internal dynamics relative to the external steering forces are explicitly taken into account. In practice, however, the ecosystem concept is used to identify more or less loosely distinguished sub-systems around some predominant feature, ranging in scale from habitat to large-scale heterogeneous sea areas. This loose definition obviously hampers a systematic approach to ecosystem comparison because, depending on the issue, the spatial borders of the ecosystem components to be compared may have to be set differently. Therefore, without some clear question identifying the aspects of ecosystems that should be compared and why, it is virtually impossible to specify appropriate system boundaries.

WGECO has been specifically asked to advise on the areas to be used in ecosystem comparisons and the meta-data available for comparisons. In the light of the foregoing, this seems a merciless task. Nevertheless, we began this process by comparing different published descriptions of different ecosystems in the North Atlantic (Section 8.1). Because the areas for comparison could not be defined, we were not able to provide recommendations regarding meta-data.

8.1 Areas to be used for ecosystem comparison

The report of the Planning Group proposed three published partitions as a basis for making spatial comparisons of ecosystems:

1) OSPAR (2000) Regions;
2) Sherman's Large Marine Ecosystems (LMEs) (Sherman and Alexander, 1989);
3) Ecological biomes/provinces (Longhurst, 1998).

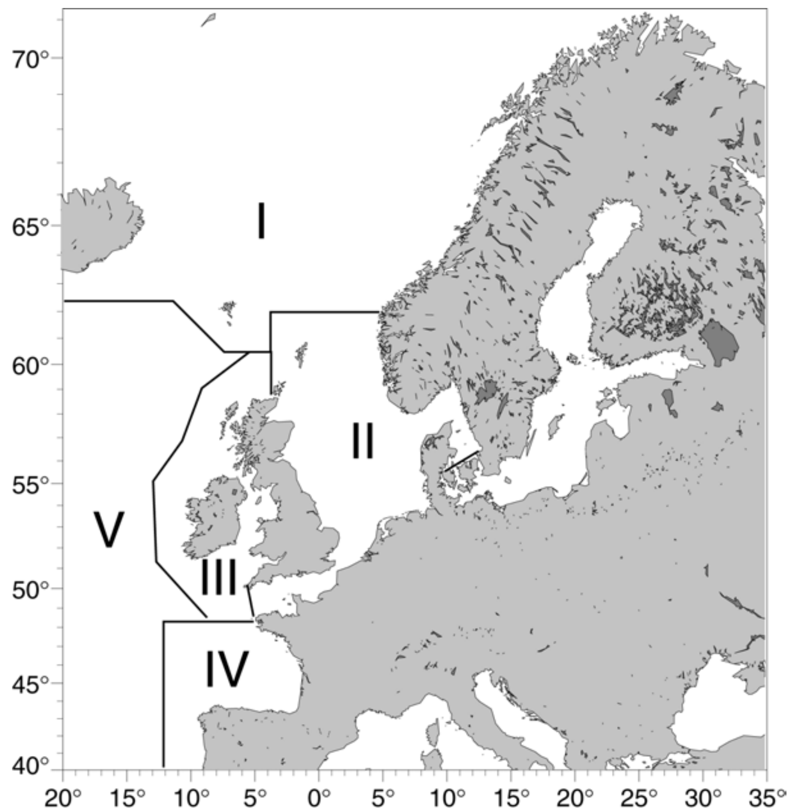
The Planning Group did not endorse any of these three, but felt that they may be useful to consider in further analysis. Most of the maps in this section were redrawn from the originals without the benefit of the exact coordinates and must be taken as approximations of the originals.

Too late in the meeting to be considered, Dinter (2001), which reviews the biogeography of the OSPAR area, was presented. It would have been valuable were time available for inclusion in this review and it is mentioned here for interested readers.

8.1.1 OSPAR regions

The OSPAR Commission decided in 1994 to subdivide the Northeast Atlantic into five regions (Figure 8.1.1.1) and prepare quality status reports (QSRs) for each which summarised their physical chemical and biological characteristics. These QSRs provide baseline information for each region, but they are rather large and may be complicated by the fact that each region may contain several heterogeneous sub-systems. The Baltic Sea is not covered by OSPAR, but is identified by HELCOM as a separate region.

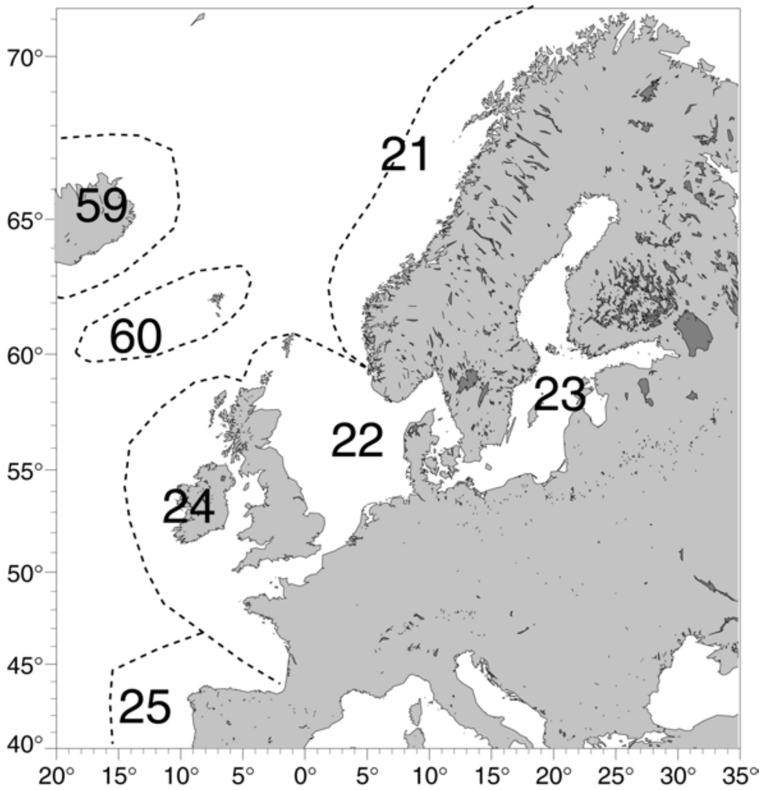
Figure 8.1.1.1. OSPAR regions: I Arctic; II Greater North Sea; III Celtic Seas; IV Bay of Biscay; and V The Wider Atlantic. Note that the Baltic is not included in this system.



8.1.2 Large Marine Ecosystems

Sherman's (1994) Large Marine Ecosystems (LMEs) take into consideration both geopolitical (man's role managing and exploiting the systems) and ecological criteria (Figure 8.1.2.1). These divisions are a bit smaller than the OSPAR regions and do not include the large deep parts of water Regions I and V. OSPAR Region I is divided into LMEs 19 (Barents Sea - not shown), 21, 59, and 60. Regions II, III and IV roughly cover the same area as LMEs 22–25 but with different partitions.

Figure 8.1.2.1. LMEs in the Northeast Atlantic: 21 Norwegian Shelf; 22 North Sea; 23 Baltic Sea; 24 Celtic-Biscay Shelf; 25 Iberian Coast; 59 Iceland Shelf; 60 Faroe Plateau.



8.1.3 Ecological biomes/provinces

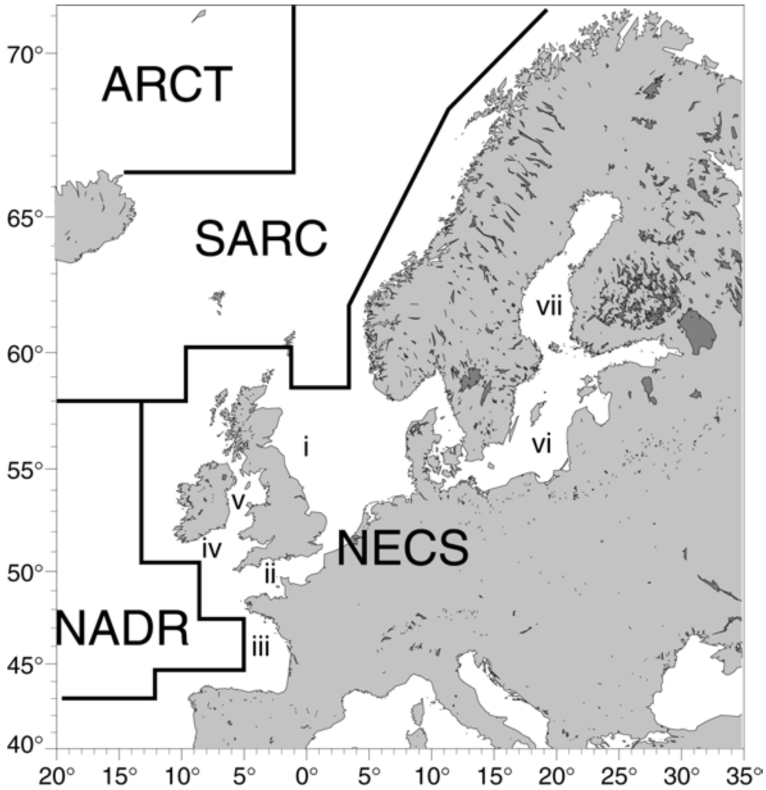
Longhurst (1998) partitioned the world's oceans into 12 biomes and further into 51 provinces based largely on oceanographic and planktonic considerations. The Northeastern Atlantic has 4 provinces (see Figure 8.1.3.1) and we focus on the Northern Atlantic Shelves Province (NECS). OSPAR I roughly corresponds to ARCT and SARC; OSPAR II and III and northern IV to NECS.

In the Report of the Planning Group on Comparing Structure of Marine Ecosystems, it was mentioned that this province could be "subdivided into seven primary divisions". These were neither specified nor shown in this report. Referring to Longhurst (1998, p. 164) we find:

> "…a first-level subdivision of the province may be useful for some purposes. If so, the following entities would probably be the classical candidates for primary divisions: (i) the North Sea from the Straits of Dover to the Shetlands; (ii) the English Channel from Dover west to Ushant; (iii) the southern outer shelf from northern Spain to Ushant, including the Acquitane and Armorican shelves off western France; (iv) the northern outer shelf including the Celtic Sea and the Irish, Malin and Hebredes shelves off Britain; (v) the Irish Sea; (vi) the Central Baltic (Gotland) Sea; and (vii) the Gulfs of Bothnia and Finland. However, there is another way of subdividing the region which is more sensitive to ecological reality."

These seven primary divisions are also shown in Figure 8.1.3.1.

Figure 8.1.3.1. Longhurst's provinces in the Northeast Atlantic: NECS Northeast Atlantic Shelves; SARC Atlantic Subarctic; ARCT Atlantic Arctic; NADR North Atlantic Drift Province. The 7 areas denoted by i–vii are what Longhurst (1998) refers to as the "classical candidates for primary divisions".



Longhurst proposes that a more ecologically based definition of subdivisions would consider depth and mixing. Further, during the winter the boundaries between mixed and thermally stratified shelf water are prominent and relatively stable. They are principally: (i) across the western entrance of the English Channel; (ii) across the northern Celtic Sea at the mouth of the Irish Sea; and (iii) within the Northern Irish Sea, and along a line from northeast England to the Friesian Coast. Thus, he proposes:

1) A central area of vertically mixed water occupying most of the English channel and the Southern North Sea (1a) as well as the central Irish Sea (1b);

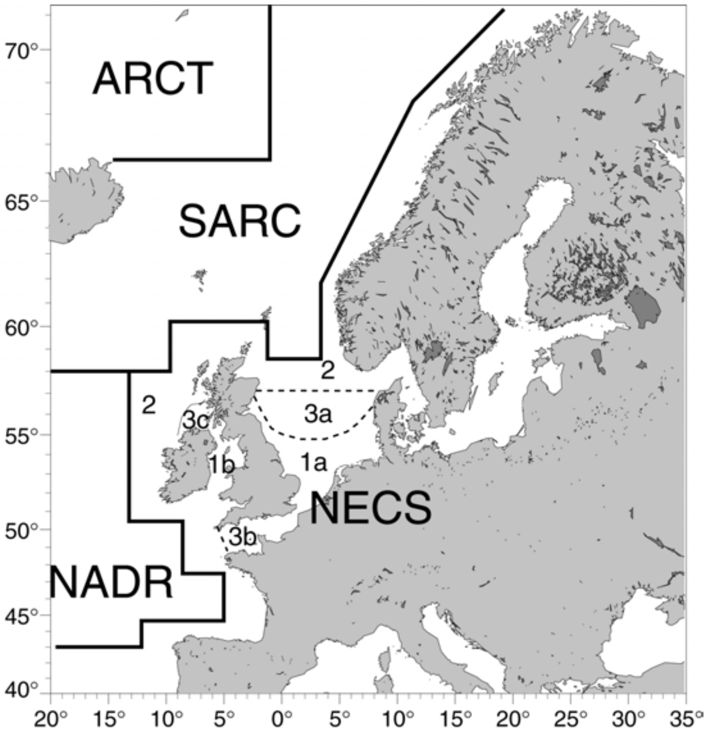
2) Stratified areas occupying the northern part of the North Sea above a line from Denmark to Yorkshire and the whole of the outer Atlantic-facing shelf from Shetland to Spain, interrupted off Ushant by an extension of No. 3 (2);

3) A transitional zone between Nos. 1 and 2 across the shelf sea fronts migrates with the tidal cycle. This occupies the western English Channel with an extension to the shelf edge off Ushant (3b), the northern Irish Sea (3c) and an arc across the southern North Sea from Yorkshire to the Dutch Coast to Denmark (3a).

The bracketed numbers refer to Figure 8.1.4.1.

Two other provinces are of interest, the North Atlantic Drift Province (NADR), which consists of deeper waters off the Shelf and the Atlantic Subarctic Province (SARC) to the north. Although Longhurst did not propose subdivision, we suggest that SARC may usefully be broken into deep and shallow waters 4 and 5.

Figure 8.1.4.1. Longhurst's provinces in the Northeast Atlantic: NECS Northeast Atlantic Shelves; SARC Atlantic Subarctic; ARCT Atlantic Arctic; NADR North Atlantic Drift Province. The 6 areas denoted by 1a,b, 2, 3a,b,c are those proposed by Longhurst (1998) as ecologically based.



8.2 Meta-data available for comparison

WGECO was not able at this meeting to provide recommendations on the meta-data available for comparisons. For discussion, a template for a table was considered, but not presented here, which had the data broken down by type and area. Before the available data by type could be compiled it is necessary to know what aspects of the ecosystems are being investigated in response to what issues. Furthermore, before the data by area could be compiled, the areas need to be defined.

8.3 Conclusions

The Planning Group did not endorse any of the 3 major area divisions, but rather stated that they all deserved further consideration. The WG had available neither data nor analytical tools to define appropriate areas for ecosystem comparison. Indeed, such definition will have to be flexible and developed iteratively as the issues and analysis progress. The chicken and egg nature of this problem is noted; once the areas are defined the data could be compiled, and once the data are assembled the areas could be refined. But the first step is the clear and structured definition of issues to be addressed.

References

Dinter, W.P. 2001. Biogeography of the OSPAR Maritime Area. BfN Federal Agency for Nature Conservation. Bonn, Germany. 165 pp.

Longhurst, A. 1998. Ecological Geography of the Sea. Academic Press. San Diego. 398 pp.

Sherman, K., and Alexander, L.M. (Eds.) 1989. Biomass yields and geography of Large Marine Ecosystems. Sustainability, biomass yields, and health of coastal ecosystems. AAAS Selected Symposium 111. Westview Press, Inc. Boulder, CO. 493 pp. (Also www.edc.uri.edu/lme)

9 PROPOSE A PROCESS TO DESCRIBE THE DISTRIBUTION OF SENSITIVE HABITATS AND MITIGATION OF FISHING IMPACTS

This term of reference reads as follows:

g) Propose a process to be able to summarise available information on the distribution of other sensitive habitats in the ICES area, and evaluate the adequacy of information as a basis for scientific advice for an "evaluation of the impact of current fishing practices on... sensitive habitats, and suggestions for appropriate mitigating measures", this should include the definition of criteria or standards for determining what is a "sensitive habitat".

9.1 Introduction

Although the ToR refers to a description of "sensitive habitats", this may also have been meant to refer to the ongoing process within ICES and OSPAR to identify and select a priority list of "threatened and declining" habitats. While the criteria used to define such threatened and declining habitats is still in the process of development within the OSPAR Biodiversity Committee (BDC) and intersessionally, these more comprehensive criteria are likely to include at least a description of the regional importance of the habitat, the rate and extent of decline, the ecological importance of the habitat, and the sensitivity and recoverability of the habitat. The Working Group chose to restrict its deliberations to aspects of habitat sensitivity (and recoverability), to evaluate existing data that may contribute to advice on the impacts of fishing activity, and to describe a procedure which could form the basis of an assessment protocol.

9.2 Defining terms

Habitat sensitivity can be defined as the degree and duration of damage caused by a standardised external factor. Sensitivity may refer to structural fragility of the entire habitat in relation to a physical impact, or to intolerance of individual species comprising the habitat to environmental factors such as exposure, salinity fluctuations or temperature variation (McDonald *et al.*, 1997).

There has been an increased interest in developing metrics to quantify habitat sensitivity, and these have largely focused on the potential impact on habitats of activities such as oil and gas exploration in coastal and intertidal environments and on post-impact mitigation. A comprehensive evaluation of species and habitat sensitivity has been developed as part of the UK MarLIN programme www.MarLIN.ac.uk (Tyler-Walters *et al.*, 2001). In general, most methods for describing habitat sensitivity are based on a logical but subjective process of allocation to different sensitivity categories. For example, Anderson and Moore (1997) applied a 4-point scale to quantify the potential consequences of oil exploration on habitats, which is weighted depending on the likelihood of occurrence. Similarly, Cooke and McMath (1998) developed a protocol for assessing sensitivity of marine benthos using the interaction between recoverability and intolerance, where "intolerance" described the inability of a species to endure damage. In both these examples, although the protocol is logical and objective, the allocation of scores is entirely subjective and there is limited guidance provided on how to implement the suggested approaches. Other coastline classification schemes have been developed which incorporate recoverability into a sensitivity scale (Gundlach and Hayes 1978), but categories are broad and rely heavily on physical rather than biological characteristics of the environment. In general, the detection of whether recovery has occurred can be problematic (McDonald and Erickson, 1994; Chapman, 1999; Underwood, 2000; Archambault *et al.*, 2001) with regard to the time-course involved and the selection of the appropriate reference and control locations. Furthermore, the level of acceptable recovery has generally not been well defined. McDonald and Erickson (1994) highlighted the advantages of using bioequivalence to test whether recovery has occurred between treated areas and reference areas, but this procedure is not common in biological sciences and should be apply more frequently.

More comprehensive map-based guidelines have been developed in the US and the Environmental Sensitivity Index (ESI) is widely used as a basis for assessment of impact of oil spills (Michel and Dahlin, 1993). The ranking of habitat sensitivity is based on features such as the slope and substrate type of the shoreline, its relative exposure to wave and tidal energy, and the productivity and sensitivity of the biological community. Although the system provides a more comprehensive use of biological characteristics than other examples, the categories are still broad. The benefit of this approach is in the production of resource maps of the coastline describing biological and physical sensitivities, and thereby providing managers with a clear and easy to use spatial tool.

Recent intersessional work in the OSPAR Biodiversity Committee has sought to progress the development of criteria for the selection of species and/or habitats that may be threatened or declining. It was agreed at the IMPACT Working Group of OSPAR in 1999 that intersessional work on the Assessment of Species and Habitats in Need of Protection should focus on testing a set of selection criteria that had already been produced at meetings in Texel (1997), Horta (1999), and further improved at IMPACT 1999 (Lindeboom and de Groot, 1998). This testing process was also thought to include the evaluation and further development of a procedure by which these Texel/Faial criteria could be applied.

This process goes further than just identifying habitat sensitivity, by incorporating a description of habitat rarity, regional importance, ecological significance and rate of decline. The way in which habitats could be allocated to a sensitivity category is unclear and insufficient guidance is provided, merely stating that it can be assessed as a function of the effect of human activity and the time taken for recovery (Table 9.2.1.a,b). It is clear from the guidance that the sensitivity of a habitat will differ according to different specific impacts of human activities and so this criterion should be applied at the end of the process. By including, and emphasising, aspects of habitat rarity and ecological significance, the influence of habitat sensitivity in the selection of threatened and declining species is effectively down-weighted. The status of habitat decline is described in terms of severe and significant decline, in relation both to extent and quality of habitat (Table 9.2.1.a).

The way in which these criteria are applied and their ultimate effectiveness depends on having a suitable habitat classification system, a sufficiently detailed habitat map, and adequate data for each habitat type so that their biological and physical characteristics can be quantified. Until each of these is in place, the selection of habitat sensitivity is likely to continue to depend on subjective assessments.

9.3 Evaluation of the potential effects of fishing activities on sensitive habitats

It was decided that the evaluation process proposed by this working group should take into account available information on what current fishing practices take place in an area, and what "sensitive habitats" exist in the same locality. It must also consider what potential effects such fishing practices might have on the sensitive habitats in terms of physical effects on the habitats themselves, and on the biological components of those habitats in terms of target and non-target species. Given the limited amount of time available to WGECO, a process is proposed that uses an existing example of habitat classification (Gubbay, 2001), fishing practices (Lindeboom and de Groot, 1998) and their impacts on sensitive habitats (Jones *et al.*, 2000, and others according to habitat and fishery type) and cross-references them in a matrix with a view to providing accessible management advice.

9.3.1 Comparing the potential impact of each fishing activity on a range of sensitive habitats

The spatial scale and magnitude of impact by fishing gears has been reviewed in previous WGECO meetings (ICES, 2000), and will not be dealt with any further here. In summary, the primary methods of fishing within the ICES area (Lindeboom and de Groot, 1998) include bottom trawling by beam trawl and otter trawl, pelagic trawling by towed gear, pelagic fishing by seine net, dredging, and the use of fixed gear such as longlines, gillnets, tangle nets and traps.

Without a comprehensive definition of habitat sensitivity, it was not considered possible, necessary, or even desirable to select a new list of sensitive habitats in the ICES area. Instead, we used the provisional list of threatened and declining habitats prepared by Gubbay (2001) and submitted to OSPAR (2001). While this may not be the most suitable or comprehensive classification, it is adopted here as an example for use in the process of presenting management advice in matrix format.

Of the types of fishing listed above, the greatest physical impact on sensitive habitats is likely to be caused by towed gears such as dredging, otter trawl and beam trawl (Collie *et al.,* 2000) through the following physical effects:

- Destruction of complex three-dimensional habitats (e.g., coral reefs/burrows/refuges) (Tuck *et al*., 1998);
- Disturbance of sediment structure (Schwinghamer *et al.*, 1996);
- Changes in topography (tracks and grooves) (Krost *et al.*, 1990);
- Resuspension of sediment/increased turbidity (clogging gills and filter-feeding animals) (Krost, 1990; Main and Sangster, 1978, 1981);
- Refluxing of chemicals (contaminants and nutrients) (Messiah *et al.*, 1991);
- Litter from abandoned/lost gear (ghost fishing).

To evaluate the adequacy of the data which describe the impact of fishing activities on sensitive habitats, as required in this ToR, we have matched the fishing practices in operation within the ICES areas against the list of sensitive habitats. These data are then reviewed in terms of their potential effect on the habitat. This has been undertaken by preparing a matrix of current fishing practices and habitat types (Table 9.3.1.1). A brief description of the effect of each gear and impact from each cell of the matrix is described in the text below. Where no impact is thought to occur, the matrix cell is marked “N/A” and no further information is supplied.

9.3.2 Mitigation measures

In cases where the impact of fishing activity is considered to be unacceptable, one management response is to implement mitigation measures to limit or eliminate the adverse effects. Such mitigation measures may range from technical measures which influence the way in which the gear operates, such as mesh size regulation or escape panels, to spatial closures to prevent access to certain fleets during part or all of the year. Different types of mitigation measure will be required depending on the sensitivity of the habitat and the fishing practice involved. ICES (2000) and Jones *et al.* (2000) have already considered mitigation measures for a number of fishing types and these include:

- Spatial closures around sensitive habitats;
- Rotation of effort from area to area;
- Modification of gear to reduce benthic impact;
- Modification of gear using biodegradable materials to prevent long-term ghost fishing;
- Restocking/reseeding – particularly of shellfish beds;
- Technical conservation measures – modification of gear to reduce by-catch in the water column;
- Legislation and enforcement to “land all catch”;
- Avoidance of areas at certain times of year when by-catch of certain species is known to exist.

Within each cell of the matrix (Table 9.3.1.1), the most appropriate mitigation measure(s) are shown, and they are briefly described in the accompanying text.

Table 9.2.1.a. Latest official version of criteria for the identification of habitats in need of protection, conservation, and where practical, restoration and/or surveillance or monitoring, otherwise known as the "Texel/Faial criteria" OSPAR 01/10/1-E. Note: these criteria are being revised by OSPAR at present.

<table>
<tr><td>1.</td><td colspan="2">Global importance (importance of the OSPAR Area for the habitat in a global context): a high proportion of the habitat occurs in the OSPAR Area.</td></tr>
<tr><td>2.</td><td colspan="2">Regional importance (importance of the sub-regions of the OSPAR Area for the habitat): a high proportion of the habitat occurs within a specific biogeographic region and/or region of national responsibility within the OSPAR Area.</td></tr>
<tr><td>3.</td><td colspan="2">Rarity: a habitat is assessed as being rare if it is restricted to a limited number of locations or to small, few and scattered locations in the OSPAR area.</td></tr>
<tr><td>4.</td><td colspan="2">Sensitivity: "very sensitive" habitat is one that is very easily adversely affected by a human activity and/or would be expected to recover only over a very long period, or not at all. A "sensitive" habitat is one that is easily adversely affected by a human activity and would be expected to recover only over a long period.
Sensitivity will be expressed in terms of:
a. impact of human activities (resistance)
b. capacity to recover (resilience), including a reflection of its degree of isolation or confinement to a small area.</td></tr>
<tr><td>5.</td><td colspan="2">Ecological significance: the habitat is very important for the wider significance of the ecological processes, functions and species that it supports.</td></tr>
<tr><td>6.</td><td colspan="2">Status of decline: Decline means a significant decline in extent or quality. The decline may be historic, recent or current. The decline can occur in the whole OSPAR maritime area or regionally.</td></tr>
<tr><td></td><td>EXTENT</td><td>QUALITY</td></tr>
<tr><td>1. Extirpated (extinct within the OSPAR Area)</td><td>A habitat which was previously present in the OSPAR Area, but no information is available that it still exists.</td><td>A habitat for which quality is affected so severely that its typical or natural components are completely destroyed.</td></tr>
<tr><td>2. Severely Declined</td><td>A habitat for which only 25 % or less of its former natural distribution in the OSPAR Area still exists. If impacts start or continue and no protection or management measures are taken, the habitat may be completely destroyed.</td><td>A habitat for which quality is negatively affected in the entire OSPAR Area so that typical or natural components can only be found in one or very few sub-regions.</td></tr>
<tr><td>3. Significantly Declined</td><td>A habitat that has declined in extent to between 25 % and 75 % of its former natural distribution in the OSPAR Area, or that has become extinct in several sub-regions.</td><td>A habitat for which quality is negatively affected by:
(1) a change of its typical or natural components over almost the entire OSPAR area, or
(2) the loss of its typical or natural components in several sub-regions.</td></tr>
<tr><td>4. Probability of significant Decline</td><td>There is a high probability that the habitat will decline by 25 % or more if no protection or management measures are taken.</td><td>There is a high probability that the habitat will significantly decline in quality if no protection or management measures are taken.</td></tr>
<tr><td colspan="3"><u>Note</u>: Lesser degrees of decline than Significantly Declined will occur but will not qualify under this criterion. Evidence for decline can be based on actual evidence or reasonable expert judgment.</td></tr>
</table>

Table 9.2.1.b. Guidance on the selection criteria for habitats in need of protection, conservation, and where practical, restoration and/or surveillance or monitoring (latest official version) OSPAR 01/10/1-E. Note: this guidance is being revised by OSPAR at present.

Criterion	**Guidance**
1.	"High proportion" is considered to be more than 75 %, when known. This criterion may require knowledge of the distribution of habitats at a global scale.
2.	"High proportion" is considered to be more than 75 %, when known.
3.	The "limited number of locations" is set at 2 % of the 50 km by 50 km UTM grid squares for each of the following three bathymetric zones: a. littoral (intertidal zone and splash zone) b. sublittoral (down to 200 m depth) c. bathyal / abyssal (below 200 m depth) The assessment is dependent on scientific judgement regarding natural abundance, range or extent and adequacy of recording.
4.	A "very long period" is considered to be more than 25 years and a "long period" in the range of 5 to 25 years, dependent on the habitat. It is considered that the sensitivity of a habitat differs according to specific impacts of different human activities and, as such, should be applied at the end of the selection process with respect to the specific impacts of human activities.
5.	Example habitats could be: spawning, breeding, reproduction, or nursery areas for fish, mammals or birds, resting and feeding areas, areas with a high natural productivity or diversity, areas with a high proportion of endemic species, and areas important as migratory routes.
6.	"Decline" will be assessed according to categories 1 to 4 described in the table below for both decline in extent and quality, recognising the following descriptions: a. Extent – based on distributional coverage or areal extent. b. Quality – judgement of decline in quality should be based on change from natural condition caused by human activities. Such judgement is likely to include aspects of biodiversity, species composition, age composition, productivity, biomass per area, reproductive ability, non-native species and the abiotic character of the habitat.

Table 9.3.1.1. Matrix of fishing gear/habitat type and mitigating measure (after ICES, 2000; Gubbay, 2001).

Fishing Activity	Sensitive Habitat Type (from Gubbay, 2001)						
	Deep-water biogenic habitats [1]	Structural benthic epifauna [2]	Benthic infauna [3]	Mollusc beds [4]	Nearshore communities [5]	Intertidal mudflats	Maerl beds
Otter trawling	AC	AC,GM	GM	AC	AC	N/A	AC
Beam trawling	N/A	AC,GM	GM	AC,GM	AC	AC	AC
Pelagic trawling	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Drift/gill netting	AC	N/A	N/A	N/A	N/A	N/A	N/A
Bottom longlining	AC?	AC,GM	N/A	N/A	N/A	AC	N/A
Pelagic longlining	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Tangle netting	AC?	GM?	N/A	N/A	AC	AC	N/A
Pot fisheries	N/A	AC,GM	N/A	N/A	AC/R	N/A	N/A
Dredging (Epibenthic)	N/A	AC	AC	AC/R	AC	AC	AC
Dredging (Hydraulic)	N/A	AC	AC	AC/R	AC	AC	N/A

Key to sensitive habitat types:

[1] Deep-water biogenic habitats: *Lophelia pertusa* reefs, carbonate mounds, oceanic ridges with hydrothermal vents, seamounts and deep-water sponge communities.

[2] Structural benthic epifauna: *Sabellaria spinulosa* reefs.

[3] Benthic infauna: Seapens and burrowing megafauna communities.

[4] Shellfish beds: *Ampharete falcata* sublittoral community, *Ostrea edulis* beds, *Modiolus modiolus* beds and intertidal mussel beds.

[5] Nearshore communities: *Zostera* beds and littoral chalk communities.

Key to mitigation measures:

AC = Area closures R = Reseeding/restocking

GM =Gear modification N/A = fishing activities thought to have no effect.

9.4 Assessment of the effects of fishing on each habitat type

9.4.1 Deep-water biogenic habitats

These habitats include any structure on the deep seabed created through biogenic means (e.g., cold-water corals, deep-water sponge communities) or natural means (e.g., carbonate mounds and mid-ocean ridges with hydrothermal vents) that act as habitats for communities. In spite of the fact that such habitats have only recently been discovered, there is abundant literature on which to base scientific advice for the evaluation of the impact of current fishing practices.

<u>Otter trawling</u>

Recent information shows that deep-water trawling does take place in areas of deep-water biogenic habitats and therefore gives rise to cause for concern (ICES, 2002). In particular, the damage to deep-water corals off the Norwegian coast with heavy gear prior to and during fishing has been described by Hall-Spencer *et al.* (2002). There is sufficient information to suggest that the most effective way of mitigating the effect of otter trawls on deep-water biogenic habitats is to close such areas to fishing.

Drift/gill netting
Evidence has been found of damage from gillnets on deep-water biogenic habitats (Fosså *et al.*, 2000), although damage is not as extensive as that caused by towed gear. The more appropriate mitigation measure is likely to be selective area closures.

Bottom longlining and tangle netting.
These fishing techniques take place in deep-water biogenic habitats for certain species (Steingrimsson, 2002) and are likely to have some effect on the habitat through entanglement and subsequent breakage of coral formations and by-catch of non-target species. It is not clear, however, that there is sufficient information available to suggest that bottom longlining or tangle netting should be excluded from deep-water biogenic habitats.

9.4.2 Structural benthic epifauna

This habitat occurs at the interface of the water column and the benthos in shallower water and includes sessile and other epibenthic organisms which form biogenic structures such as *Sabellaria spinulosa* reefs and sponges. The main threat to such habitats comes from towed gear, such as trawls and dredges which physically damage the habitats and destroy the biogenic structures created by their inhabitants.

Otter trawling
Otter trawling has an adverse impact on structural benthic epifauna (Dayton *et al.,* 1995; Engel and Kvitek, 1998; Prena *et al.,* 1999; ICES, 2000; Linnane *et al.,* 2000). Effects can be mitigated by spatial closures where the habitat is considered sensitive, and by temporal closures where the habitat is considered more robust to allow the fauna time to recover. Gear modification may also mitigate direct impact to these habitats.

Beam trawling
There is evidence (Lindeboom *et al.*, 2000; Kaiser *et al.*, 1996b, 1996c, 1998a; Lindeboom and de Groot, 1998; Freese *et al.*, 1999) that beam trawling has a more profound effect upon the benthos, in terms of disturbance, displacement and destruction, than otter trawling per unit area of impact (Philippart, 1996, 1998). Effects can be mitigated in the same way as otter trawling, with more emphasis on spatial closures where habitats are considered to be sensitive or slow to recover.

Bottom longlining
There is anecdotal evidence to suggest that, while bottom longlining will not have as profound an effect on biogenic habitats as trawling, the potential exists for some damage through entanglement or "ghost fishing", although this has been difficult to reference. It is therefore suggested that spatial and temporal closures, and gear modification, should only be considered where the habitat is proven to be sensitive and slow to recover from damage.

Tangle netting
It can be suggested that tangle netting will cause disruption to structural benthic epifauna habitats through entanglement with structures and subsequent breakage, although once again, convincing information to support this suggestion is lacking. Mitigation measures could be introduced by spatial and temporal closures where the habitat is known to be particularly sensitive and through modification of the gear to prevent "ghost fishing".

Pot fisheries
While there is little published literature on the effect of pot fisheries on this kind of habitat, the most likely damage to epibenthic structures would be through some limited physical damage from the placement of the gear itself. Mitigation measures could be introduced such as temporal closures where the habitat is known to be particularly sensitive and through modification of the gear to prevent "ghost fishing", although priority is likely to be low.

Dredging (Epibenthic)
There is evidence (ICES, 2000; Fox *et al.*, 1996; Linnane *et al.*, 2000; Thrush *et al.*, 1995; Kaiser *et al.*, 1998b; Turner *et al.,* 2000; Veale *et al.*, 2000) to suggest that epibenthic dredging would damage biogenic structures on the seabed. It is suggested that epibenthic dredging should be restricted by spatial closure from structural benthic epifauna habitats that are known to be sensitive.

Dredging (Hydraulic)
Since hydraulic dredging is even more likely to impact negatively on structural epibenthic communities, it should be restricted by spatial closure of structural epifauna habitats that are known to be sensitive.

9.4.3 Benthic infauna

This habitat comprises the sediment of the seabed and communities of such burrowing animals as seapens, *Spisula*, razor clams and other burrowing megafauna communities (Hughes, 1998). It is reasonable to assume that any fishing activity that disturbs the seabed will impact on this habitat.

<u>Otter trawling</u>
There is published evidence to demonstrate the effect of otter trawling on benthic infauna (Engel and Kvitek, 1998; Gilkinson *et al.*, 1998). Mitigation may be carried out by spatial closures in areas of high sensitivity, and by temporal closures and gear modification in areas where habitats are more robust. One innovative technical conservation measure includes the modification of the tickler chain of an otter trawl with "roller-balls" (Linnane *et al.*, 2000).

<u>Beam trawling</u>
A number of studies have shown an impact of beam trawling on this habitat (Bergman and Hup, 1992; Kaiser and Spencer, 1996; Kaiser *et al.*, 1996a, 1996b), and the impact of this gear on benthic infauna is thought to be greater than that of otter trawling. In areas where habitats are considered highly sensitive, the effect should be mitigated by spatial closures, with temporal closures in habitats where recovery is more likely.

<u>Dredging (Epibenthic)</u>
This type of fishing, while specifically designed to target the epibenthos, will inevitably have an effect on benthic infauna (Currie and Parry, 1996, 1999; Hill *et al.*, 1999; Kaiser *et al.,* 1998b) through damage to filter feeding mechanisms (seapens), siphons (in bivalve molluscs) and possibly through disruption of habitat integrity. Effects can be mitigated by spatial closures in areas of high sensitivity and by temporal closures in areas where recovery is thought to be faster.

<u>Dredging (Hydraulic)</u>
Since this gear is specifically designed to target benthic infauna (e.g., razor clams and other burrowing molluscs), it will have one of the largest effects of all the types of fishing gear used on infaunal communities (Hall *et al.*, 1990; Dayton *et al.*, 1995; Tuck *et al.*, 1999). The effect should be mitigated by spatial closures.

9.4.4 Mollusc beds

This habitat comprises mollusc beds (intertidal mussels, oysters and horse mussels) that are considered at risk from fishing activities. They are primarily located close to land in shallow water and, as well as fishing, are threatened also by pollution and suspended sediment load from the shore.

<u>Otter trawling/Beam trawling</u>
Physical disturbance to the benthos and breakage of animal shells are the major impacts of these two fishing activities on mollusc beds (Hoffmann and Dolmer, 2000; Witbraard and Klein, 1994). Spatial and/or temporal closure and modification of the fishing gear are potential mitigation measures which can be applied to minimise impact.

<u>Dredging (Epibenthic)</u>
As bivalve molluscs are harvested by epibenthic dredges, there is no shortage of available information which describes the impact of dredging on these habitats (Auster *et al.*, 1996; Jennings and Kaiser, 1998; Watling and Norse, 1998; Bradshaw *et al.*, 2000). Information also confirms that closure and reseeding of shellfish grounds for the target species involved is a workable mitigating measure that should be promoted. The seeding of scallop beds is well established for some parts of the east coast of Canada (P. Archambault, pers. comm.), as well as in Ireland and in Scotland.

<u>Dredging (Hydraulic)</u>
Substantial information also exists on the effect of hydraulic dredging on mollusc beds. Chevarie *et al.* (2001) observed that hydraulic dredges affect juvenile and adult abundance at certain crucial times of year, making it imperative to select which times to operate dredges so as to minimise damage. Rotation or modification of the date of collecting the target species should be set to occur before recruitment events to minimise the effect on spatfall. Literature on this subject suggests a relatively fast recovery or a low impact (Hall *et al.,* 1990). Reseeding could be a mitigation measure for the target species but the result of such mitigation is not well understood.

9.4.5 Nearshore communities

These habitats (*Zostera* communities, littoral chalk communities) comprise species in shallow water that are considered under threat (Birkett *et al.*, 1998b; Holt *et al.*, 1998).

Otter trawling, Beam trawling, Tangle netting
The impact of beam trawls, otter trawls and tangle netting on these habitats is not well understood and there are few published descriptions. In sensitive inshore habitats, which are vulnerable to the scouring effect of otter and beam trawls, spatial and temporal closures may be appropriate mitigating measures.

Pot fisheries
The primary impact of pot fisheries is to remove the target species, as well as some by-catch. Mitigation measures such as spatial closure or restocking may be appropriate. In general, pot fisheries have limited impacts on habitats.

Dredging (Epibenthic)/Dredging (Hydraulic)
Dredging a sensitive nearshore community will produce a number of negative effects. Any biological damage to *Zostera* beds will reduce the surface area for attachment by the early juvenile stage of scallop and other invertebrates (Fonseca *et al.*, 1984) which can be mitigated by spatial closures. Such measures are applicable to any sensitive nearshore habitat.

9.4.6 Intertidal mudflats

While not normally considered as being at risk from fishing activities, these habitats are an important feeding ground for shorebirds. Such habitats are coming under increasing pressure from bait digging, coastal construction and other human activities related to fishing (Elliot *et al.*, 1998).

Beam trawling
The impact of beam trawling on intertidal mud flats, which takes place for shrimp and flatfish, will depend on the penetration depth of the gear and the degree to which the habitat is already affected by natural disturbance, which in turn will vary with the time of year (Kaiser *et al.*, 1996a).

Bottom longlining/tangle netting
While it is known that static gear, such as longlines and tangle nets, are used on intertidal mud flats to catch flatfish, their effect on the habitat will be less than with active fishing methods such as trawling and dredging.

Dredging (Epibenthic)/Dredging (Hydraulic)
Negative impact on the benthic community of intertidal mudflats has been observed but not on a long-term basis (Kaiser *et al.*, 1996a). Spatial closure of an area is a possible and appropriate mitigating measure. The impact of fishing activities largely depends on the sediment type, since communities in mobile and coarser sediment are less likely to be disturbed (Moore, 1991). Kaiser *et al.* (1998c) suggested that, if the mechanism for recolonisation is by larval settlement, a restriction on harvesting to early winter may encourage site restoration.

9.4.7 Maerl beds

Mearl beds are large aggregations of calcareous algae and are under threat primarily from dredging activities as a source of raw material for pharmaceutical and industrial use. They support very high species diversity and are slow-growing in European waters. They are very sensitive compared to other sedimentary bottom types (Birkett *et al.*, 1998a; Hall-Spencer and Moore, 2000). Any potential impact from fishing activities, therefore, gives cause for concern.

Otter trawling, Beam trawling, Dredging (Epibenthic)
These three kinds of fishing activities can have profound and long-term impacts on maerl habitats. These impacts were still present four years after the experiment. Some literature on the sensitivity of maerl beds related to fishing impact is available in Grall and Glémarec (1998), and Hall-Spencer and Moore (2000) and this suggests a decrease of > 70 % of live maerls after scallop dredging. No information seems available to suggest what mitigation measures might be taken, but the high sensitivity of these habitats, and a decrease in their abundance at least along the Scottish coast, suggests that the most effective mitigation measure should be a spatial closure. Otter trawling has a less negative impact on the Maltese mearl bed than scallop dredging (BIOMAERL team, 1999).

9.5 Assessment of the matrix approach

From the above material, it is clear that:

- More work is required to properly define "sensitive habitats", and in this way to identify environments which require genuine management action.

- In general, sufficient information exists in the scientific literature to predict the effects of the majority of existing fishing practices on a number of habitats that may be considered as proxies for sensitive habitats, and to suggest mitigating actions. Gaps mainly exist in relation to the effects of bottom longlining and tangle netting, and the type of mitigation measures that may be appropriate.
- There was not sufficient opportunity to consider the location of such habitats, although it is suggested that detailed spatial mapping of sensitive habitats could be a next step towards ecosystem-based management.
- The matrix approach developed here, matching fishing activity to habitat type and mitigating measures, could be developed further as a computer-based model (perhaps through GIS) for use as a Marine Information System management tool.
- Ghost fishing by fixed nets and longlines was not included as a usual practice of fishing activity, but under certain circumstances could have profound effects on non-target species such marine mammals (Dayton *et al.*, 1995), crabs (Breen, 1987), gadoids and crustacea (Kaiser *et al*., 1996d). Evidence of ghost fishing on deep-water biogenic habitats by deep-water longlines and gillnets has already been demonstrated (ICES, 2002), but ghost fishing will occur through pot fisheries, where lost, but unmodified, pots may continue fishing for crabs, lobster and whelks for a long period of time. The literature is scarce on this subject (Eno *et al.*, 1996), but Breen (1987) reports that 11 % of crab pots in the Fraser Estuary district (BC, Canada) are lost, which could continue to fish up to 7 % of the biomass reported in this area. When fitted with biodegradable panels the gear stops fishing after a period of time, and this procedure is widely applied in some areas. Further technological advances with biodegradable fishing materials used in other gears would also be a useful contribution.

9.6 Developing the process

One of the most important parts of this ToR was the development of a process by which the effect of fishing activity on habitat sensitivity could be evaluated, and appropriate mitigation measures put in place.

WGECO proposes to use a decision tree of the sort described in Figure 9.6.1.a as our description of the most suitable and pragmatic means for developing this process. This has been based on progress made elsewhere, and draws on detailed decision-making processes described by Hiscock *et al.* (1999), Tyler-Walters *et al.* (2001) and Collie *et al.* (2000). In the procedure described here, fishing activity has a range of impacts on the seabed, and the sensitivity of the habitat to the specific type of impact will result in a range of different responses. If the habitat is not sensitive to the impact, i.e. the activity does not have a detectable effect on the structure and functioning of the habitat or on the viability or survival of important or dominant species, then there is clearly no management action that need be taken. Some habitats are likely to show a low level of sensitivity where species are unlikely to be damaged, but the viability of a community may be reduced. The next higher level of sensitivity may result in a habitat being degraded to some extent and the diversity and functionality of the habitat reduced. Under both these scenarios the level of change in the habitat will require the implementation of a surveillance programme to ensure that further degradation does not take place, but may not be so severe that mitigation measures are required to limit the impact of fishing activity (Figure 9.6.1.a). Other scenarios of fishing impact may affect habitats of intermediate or high sensitivity, where the populations of important/dominant species in a habitat are likely to be killed, the viability of these populations reduced and the habitat partially or completely destroyed. Under these conditions it is necessary to impose mitigation measures to limit the site-specific impact, and to ensure that there is feedback so that no future impacts of this sort occur (Figure 9.6.1.a).

9.6.1 Incorporating ICES advice into this process

While this may be a pragmatic approach, it is not complete without a clear indication of how information and advice on each of these levels can be provided. There are a number of stages at which specific ICES working groups and the ICES advisory system can contribute to this process, and these are illustrated in Figure 9.6.1.b.

There is considerable experience in the assessment working groups of the fleet distribution and effort of the major European bottom-trawl fisheries (WGNSDS; WGNSSK; WGSSDS), and several WGECO ToRs have addressed the impact of fishing activity on the marine ecosystem (ICES, 1996, 2000). Despite this, there is still an urgent need for higher-resolution spatial data sets, such as those already collected for some parts of the fleet using satellite monitoring, and for those data sets to be made more widely available. It is only these data that will allow us to advise on the spatial extent of fishing activity, and the potential impacts of these gears on demersal habitats.

Further work is needed on the sensitivity of habitats to various impacts. Rapid progress on this topic can and should be made by the expert groups, Working Group on Marine Habitat Mapping (WGMHM) and Benthos Ecology Working Group (BEWG), with input from WGECO. Evaluation and definition of appropriate levels of habitat sensitivity may also come from these groups, but should also be considered in the future work programme of WGECO. We expect that progress with this work will also be generated within OSPAR and during their intersessional work.

Further work is also needed on the recovery rates of different habitats after impact, and although some useful progress has been reported by the Working Group on the Effects of Extraction of Marine Sediments on the Marine Ecosystem (WGEXT) in relation to the impacts of marine aggregate extraction on benthic habitats, additional research is required to develop this advice in relation to fishing impacts. It may be appropriate to make use of the expertise of specialist groups such as BEWG and WGMHM in order to further develop these habitat-specific issues (Figure 9.6.1.b).

The development of surveillance and/or monitoring programmes is an integral part of the current respective EEA and OSPAR initiatives to identify indicators and quality objectives for each aspect of the marine ecosystem. It is essential that long-term monitoring programmes have clear objectives so that data are assessed in relation to pre-defined criteria, and there are clear opportunities for WGSAEM for further developing appropriate and robust statistical protocols. There may also be opportunities for SGEAM to contribute to the development of ecosystem monitoring issues.

Mitigating the potential impacts of fishing activity has been addressed here (Section 9.4), and dealt with comprehensively in earlier reports of this working group (Table 6.4.4.1 in ICES, 2000). There is sufficient understanding of the spatial and temporal aspects of fishing impacts to suggest broad categories of mitigation, and this could be readily transmitted via WGECO to ACE (Figure 9.6.1.b).

Figure 9.6.1.a. Potential decision tree relating the impact of fishing activity on habitats to management advice.

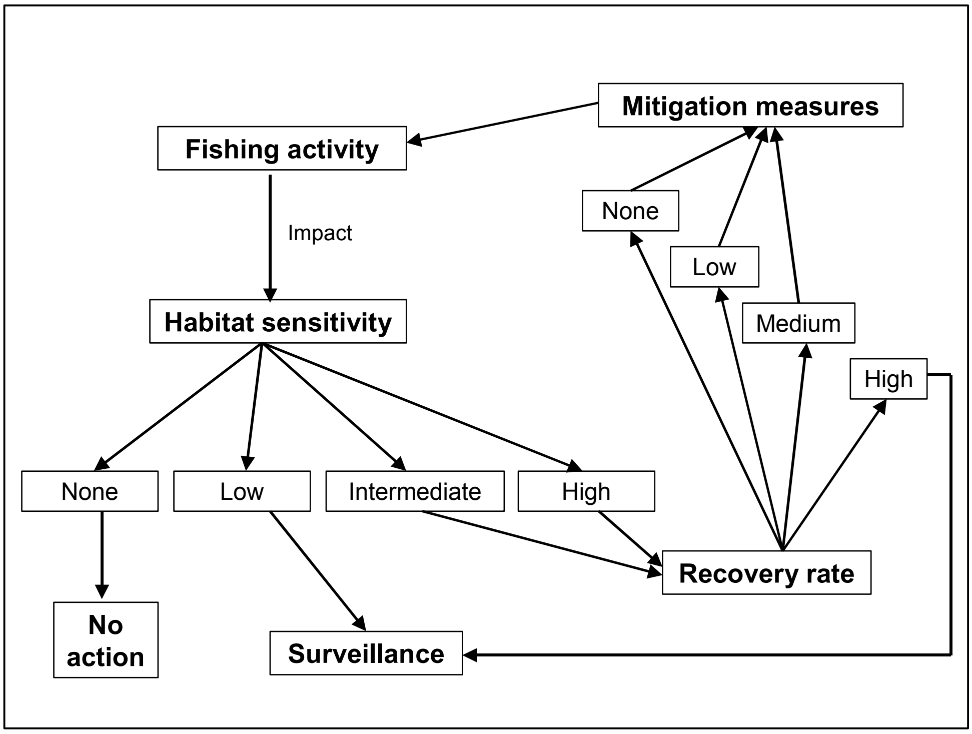
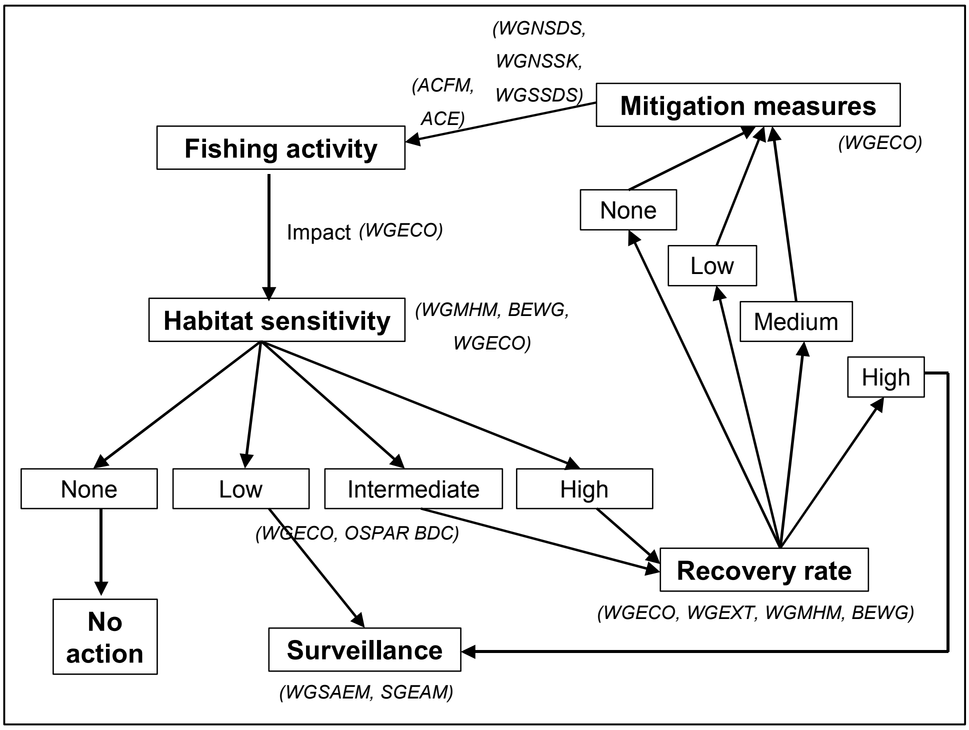


Figure 9.6.1.b. Suggested contributions by ICES advisory bodies and working groups, etc., to the provision of data and advice on issues of habitat sensitivity.



9.7 Summary conclusions

- The matrix approach to describing the potential impact of fishing gears on a selection of sensitive habitats identified that the next step is to provide spatially referenced data from habitat mapping programmes.
- The decision-tree approach was recommended as an effective way of implementing a process which mitigates the impact of fishing gears on specific habitats. This requires information on the range of fishing activities taking place, the sensitivity of habitats on which those activities occur, their recovery rates, and appropriate mitigation measures that can be put in place.
- The ICES advisory system can contribute to this process at a number of stages, and Figure 9.6.1.b illustrates the points at which ICES advisory groups and working groups can be most effective.
- The sensitivity and recovery rate of habitats are poorly understood, and further detailed evaluation needs to be undertaken in BEWG and WGMHM. Without this work, the selection of the most appropriate mitigation will be delayed.
- Some long-term data sets provide an excellent means of monitoring change, but these evaluations must be based on a statistically robust protocol with clear hypotheses.

9.8 Recommendations

- WGMHM and BEWG should progress work on the sensitivity of marine habitats to fishing impacts with input from WGECO.
- WGECO should include evaluation and definition of appropriate levels of marine habitat sensitivity in future work plans along with work on the recovery rates of different habitats after impact from fisheries activities.
- WGSAEM and SGEAM should work together to develop appropriate and robust statistical protocols for ecosystem surveillance and monitoring programmes as an integral part of the current respective EEA and OSPAR initiatives to identify indicators and quality objectives for each aspect of the marine ecosystem.
- The broad categories of mitigation suggested by this study, and dealt with comprehensively in earlier reports of this working group (Table 6.4.4.1 in ICES, 2000), should be transmitted via WGECO to ACE.

References

Anderson, S., and Moore, J. 1997. Guidance on assessment of seabed wildlife sensitivity for marine oil and gas exploration. A report to JNCC from OPRU, Neyland, UK. Report No. OPRU/18/96.

Archambault, P., Banwell, K., and Underwood, A.J. 2001. Temporal variation in the structure of intertidal assemblages following the removal of sewage. Mar. Ecol. Prog. Ser., 222: 51–62.

Auster, P. J., Malatesta, R. J., Langton, R.W., Watling, L., Valentine, P.C., Donalson, C.L.S., Langton, E.W., Shepard, A.N., and Babb, I.G. 1996. The impacts of mobile fishing gear on seafloor habitats in the Gulf Maine (Northwest Atlantic): implications for conservation of fish populations. Reviews in Fisheries Science, 4: 185–202.

Bergman, M. J. N., and Hup, M. 1992. Direct effects of beam trawling on macrofauna in a sandy sediment in the southern North Sea. ICES J. Mar. Sci., 49: 5–11.

BIOMAERL team. 1999. BIOMAERL: maerl biodiversity functional structure and anthropogenic impacts. EC Contract No. MAS3-CT95-0020, 973 pp.

Birkett, D. A., Maggs, C. A., and Dring, M. J. 1998a Maerl (Volume V). An overview of dynamics and sensitivity characteristics conservation management of marine SACs. Scottish Association for Marine Science (UK Marine SACs Project).

Birkett, D. A., Maggs, C. A., Dring, M. J., Boaden, P. J. S., and Seed, R. 1998b. Infralittoral reef biotopes with kelp species (Volume VII). An overview of dynamics and sensitivity characteristics conservation management of marine SACs. Scottish Association for Marine Science (UK Marine SACs Project).

Bradshaw, C., Veale, L.O., Hill, A.S., and Brand, A.R. 2000. The effects of scallop dredging on gravelly seabed communities. *In* Effects of fishing on non-target species and habitats: biological conservation and socio-economic issues, pp. 105–117. Ed. by M. J. Kaiser and S. J. de Groot. Blackwell Science, Oxford. 399 pp.

Breen, P.A. 1987. Mortality of Dungeness crabs caught by lost traps in the Fraser Estuary, British Columbia. N. Am. J. Fish. Mont., 7: 429–435.

Chapman, M. G. 1999. Improving sampling designs for measuring restoration in aquatic habitats. J. Aquat. Ecosyst. Stress Recov., 6: 235–251.

Chevarie, L., Myrand, B., Archambault, P. and Provencher, L. 2001. Étude d'impacts d'un engin hydraulique pour la récolte de myes (*Mya arenaria*) dans la lagune du Havre-aux-Basques, été et automne 2000. SODIM, Québec, Canada.

Collie, J. S., Hall, S. J., Kaiser, M. J., and Poiner, I. R. 2000. A quantitative analysis of fishing impacts on shelf-sea benthos. J. Anim. Ecol., 69: 785–798

Cooke, A., and McMath, M. 1998. SENSMAP: Development of a protocol for assessing and mapping the sensitivity of marine species and benthos to maritime activities. CCW marine Report: 98/6/1.

Currie, D. R., and Parry, G. D. 1996. Effects of scallop dredging on a soft sediment community: a large-scale experimental study. Mar. Ecol. Prog. Ser. 134: 131–150.

Currie, D. R., and Parry, G. D. 1999. Impacts and efficiency of scallop dredging on different soft substrates. Can. J. Fish. Aquat. Sci., 56: 539–550.

Davison, D. M., and Hughes, D. J. 1998. *Zostera* biotopes (Volume I). An overview of dynamics and sensitivity characteristics conservation management of marine SACs. Scottish Association for Marine Science (UK Marine SACs Project).

Dayton, P. K., Thrush, S. F., Agardi, M. T., and Hofman, R. J. 1995. Environmental effects of marine fishing. Aquatic conservation: marine and freshwater ecosystems, 5: 205–232.

de Groot, S.J., and Apeldoorn, J., 1971. Some experiments on the influence of the beam trawl on the bottom fauna. ICES CM 1971/B:2.

Elliot, M., Nedwell, S., Jones, V., Read, S.J., Cutts, N.D., and Hemingwat, K. L. 1998. Intertidal sand and mudflats and subtidal mobile sandbanks (Volume II). An overview of dynamics and sensitivity characteristics conservation management of marine SACs. Scottish Association for Marine Science (UK Marine SACs Project).

Engel, J., and Kvitek, R. 1998. Effects of otter trawling on benthic community in Monterey Bay National Sanctuary. Cons. Biol., 12: 1204–1214.

Eno, N. C., MacDonald, D.S., and Amos, S. C. 1996. A study on the effects of fish (crustacea/molluscs) traps on benthic habitat and species. Report to the European Commission.

Fonseca, M. S., Thayer, G. W., Chester, A. J., and Foltz, C. 1984. Impact of scallop harvesting on eelgrass (*Zostera marina*) meadows: implication for management. N. Am. J. Fish. Mont., 4: 286–293.

Fosså, J.H., Moertensen, P.B., and Furevik, D.M. 2000. *Lophelia*-korallrev langs Norskekysten forekomst og tilstand. Kisken og Havel 2–2000. Havforskningsinstituttenet, Bergen.

Fox, G.M., Ball, B.J., Munday, B.W., and Pfeiffer, N. 1996. The IMPACT II study: preliminary observations on the effects of bottom trawling on the ecosystem of the *Nephrops* grounds in the N.W. Irish Sea. *In* Irish Marine Science 1995, pp. 337–354. Ed. by B.F. Keegan and R. O'Connor. Galway University Press Ltd., Galway, 626 pp.

Freese, L., Auster, P., Heifetz, J. and Wing, B. 1999. Effects of trawling on seafloor habitat and associated invertebrate taxa in the Gulf of Alaska. Mar. Ecol. Prog. Ser., 182: 119–126.

Gilkinson, K., Paulin, M., Hurley, S. and Schwinghamer, P. 1998. Impacts of trawl door scouring on infaunal bivalves: results of a physical trawl door model/dense sand interaction. J. Exp. Mar. Biol. Ecol., 224: 291–312.

Grall, J., and Glémarec, M. 1998. Biodiversité des fonds de maerl en Bretagne: approche fonctionelle et impacts anthropiques. Vie et Milieu, 47: 339–349.

Gubbay, S. 2001. Review of proposals for an initial list of threatened and declining species in the OSPAR maritime area. Report for National Institute for Coastal and Marine Management, Ministry of Transport, Public Works and Water Management, The Netherlands.

Gundlach, E.R. and Hayes, M.O. 1978. Classification of coastal environments in terms of potential vulnerability to oil spill damage. Marine Technical Society Journal, 12(4): 18-27.

Hall, S. J., Basford, D. J. and Robertson, M. R. 1990. The impact of hydraulic dredging for razor clams *Ensis* sp. on an infaunal community. Neth. J. Sea Res. 27: 119-125.

Hall-Spencer, J. M., and Moore, P. G. 2000. Impact of scallop dredging on maerl grounds. *In* Effects of fishing on non-target species and habitats: biological conservation and socio-economic issues, pp. 105–117. Ed by M. J. Kaiser and S. J. de Groot. Blackwell Science, Oxford, 399 pp.

Hall-Spencer, J., Allain, V., and Fosså, J.H. 2002. Trawling damage to NE Atlantic ancient coral reefs. Proceedings of the Royal Society of London, B. Online paper 01PB0637.

Hill, B.J., and Wassenberg, T. J. 1990. Fate of discards from prawn trawlers in Torres Strait. Australian Journal of Marine Science, 41: 53–64.

Hill, A., Veale, L. Pennington, D., Whyte, S., Brand, A., and Hartnoll, R. 1999. Changes in Irish Sea benthos: possible effects of 40 years of dredging. Estuar. Coast. Shelf Sci., 48: 739–750.

Hiscock, K., Jackson, A., and Lear, D. 1999. Assessing seabed species and ecosystem sensitivities: existing approaches and developments. Report to the Department of Environment, Transport and the Regions from the Marine Life Information Network (MarLIN), Marine Biological Association, Plymouth, UK. MARLIN report 1. June 2001.

Hoffmann, E., and Dolmer, P. 2000. Effect of closed areas on distribution of fish and epibenthos. ICES J. Mar. Sci., 57: 1310–1314.

Holt, T. J., Rees, E. I., Hawkins, S. J., and Seed, R. 1998 Biogenic reefs (Volume IX). An overview of dynamics and sensitivity characteristics conservation management of marine SACs. Scottish Association for Marine Science (UK Marine SACs Project).

Hughes, D. J. 1998. Sea pens and burrowing megafauna (Volume III). An overview of dynamics and sensitivity characteristics conservation management of marine SACs. Scottish Association for Marine Science (UK Marine SACs Project).

ICES. 1995. Report of the Baltic Salmon and Trout Assessment Working Group. ICES CM 1995/Assess:16.

ICES. 1996. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 1996/Assess/Env:1.

ICES. 1997. Report of the Study Group on Unaccounted Mortalities in Fisheries. ICES CM 1997/B:1.

ICES. 2000. Report of the Working Group on the Ecosystem Effects of Fishing Activities. ICES CM 2000/ACME:02

ICES. 2002. Report of the Study Group on Mapping the Occurrence of Cold Water Corals. ICES CM 2002/ACE:05

Jennings, S., and Kaiser, M. J. 1998. The effects of fishing on marine ecosystems. Adv. Mar. Biol., 34: 201–352.

Jones, L.A., Hiscock, K., and Connor, D.W. 2000. Marine habitat reviews. A summary of ecological requirements and sensitivity characteristics for the conservation and management of marine SACs. Joint Nature Conservation Committee, Peterborough, UK (UK Marine SACs Project report).

Kaiser, M. J. and Spencer, B. 1996. The effects of beam-trawl disturbance on infaunal communities in different habitats. J. Ani. Ecol., 65: 348–358.

Kaiser, M. J., Edwards, D. B., and Spencer, B.E. 1996a. Infaunal community changes as a result of commercial clam cultivation and harvesting. Aquat. Living Resour., 9: 57–63.

Kaiser, M. J., Hill, A., Ramsay, K., Spencer, B. E., Brand, A., Veale, L., Prudden, K., Rees, E. I. S., Munday, B. W., Ball, B. and Hawkins, S. 1996b. Benthic disturbance by fishing gear in the Irish Sea: a comparison of beam trawling and scallop dredging. Aquatic Cons. Mar. Fresh. Ecosyst., 6: 269–285.

Kaiser, M. J., Ramsay, K., and Spencer, B. E. 1996c. Short-term ecological effects of beam trawl disturbance in the Irish Sea: a review. ICES CM 1996/Mini:5.

Kaiser, M. J., Bullimore, B., Newman, P., Lock, K., and Gilbert, S. 1996d. Catches in ghost fishing set nets. Mar. Ecol. Prog. Ser., 145: 11–16.

Kaiser, M. J., Edwards, D., Armstrong, P., Radford, K., Lough, N., Flatt, R. and H. Jones. 1998a. Changes in megafaunal benthic communities in different habitats after trawling disturbance. ICES J. Mar. Sci., 55: 353–361.

Kaiser, M. J., Armstrong, P., Dare, P., and Flatt, R. 1998b. Benthic communities associated with a heavily fished scallop ground in the English Channel. J. Mar. Biol. Assoc., 78: 1045–1059.

Kaiser, M.J., Laing, I., Utting, S.D., and Burnell, G.M. 1998c. Environmental impacts of bivalve mariculture. J. Shellfish Res., 17: 59–66.

Kirchhoff, K. 1982. Wasservogelverluste durch die Fischerei and der Schleswig-Holsteinichen Ostseeküste. Die Vogelwelt, 103: 81–89.

Krost, P. 1990. The impact of otter-trawl fishery on nutrient release from the sediment and macrofauna of Kieler Bucht (Western Baltic). Berichte aus dem Institut für Meereskunde an der Christian-Albrechts-Universität, Kiel. Nr. 200: 157 pp. (in German, English summary).

Krost, P., Bernhard, F., Werner, F., and Hukreide, W. 1990. Otter trawl tracks in Keil Bay (Western Baltic) mapped by side-scan sonar. Meersforschung, 32: 344–354.

Lindeboom, H.J. 2000. The need for closed areas as conservation tools. *In* Effects of Fishing on non-target species and habitats: biological conservation and socio-economic issues. Ed. by M.J. Kaiser and S.J. de Groot, pp 290–30. Blackwell Science.

Lindeboom, H. J. and de Groot, S. J. 1998. The effects of different types of fisheries on the North Sea and Irish Sea benthic ecosystems. RIVO-DLO report C003/98.

Linnane, A., Ball, B., Munday, B., van Marlen, B., Bergman, M., and Fonteyne, R. 2000. A review of potential techniques to reduce the environmental impact of demersal trawls. Irish Fisheries Investigations (New Series) No. 7. Marine Institute, Dublin.

Main, J., and Sangster, G.I. 1978. The value of direct observation techniques by divers in fishing gear research. Scottish Fisheries Research Report No. 12, Dept. of Agriculture and Fisheries for Scotland, Aberdeen, Scotland. 15 pp.

Main, J., and Sangster, G.I. 1981. A study of sand clouds produced by trawl boards and their possible effect on fish capture. Scottish Fisheries Research Report No. 20, Dept. of Agriculture and Fisheries for Scotland, Aberdeen, Scotland. 19 pp.

McDonald, L. L., and Erickson, W. P. 1994 Testing for bioequivalence in field studies: has a disturbed sites been adequately reclaimed? *In* Statistics in ecology and environmental monitoring, pp. 183–197. Ed. by D. J. Fletcher and B. F. J. Manly. University of Otago Press, Dunedin.

McDonald, D.S., Little, N., Eno, C., and Hiscock, K. 1997. Towards assessing the sensitivity of benthic species and biotopes in relation to fishing activities. Aquatic Conservation: Marine and Freshwater Ecosystems, 6.

Messiah, S.N., Rowell, T.W., Peer, D.L., and Cranford, P.J. 1991. The effects of trawling, dredging and ocean dumping on the eastern Canadian continental shelf seabed. Continental Shelf Research, 11: 1237–1263.

Michel, J., and Dahlin, J. 1993. Guidelines for developing digital environmental sensitivity index atlases and databases. Research Planning Inc. 1998. Environmental Sensitivity Index (ESI). http://www.researchplanning.com/esi/esi.htm.

Moore, J. 1991. Studies on the impact of hydraulic cockle dredging on intertidal sediment flat communities. Final report.

OSPAR 2001. Summary Record of the Workshop on Threatened and Declining Species and Habitats. OSPAR Commission, London, Doc. BDC 01/4/2-E.

Philippart, C. J. M. 1996. Long term impact of bottom fisheries on several bycatch species of demersal fish and benthic invertebrates in the southeastern North Sea. ICES CM 1996/Mini 6.

Philippart, C. J. M. 1998. Long-term impact of bottom fisheries on several by-catch species of demersal fish and benthic invertebrates in the south-eastern North Sea. ICES J. Mar. Sci., 55: 342–352

Prena, J., Schwinghamer, P., Rowell, T., Gordon, D. J., Gilkinson, K., Vass, W., and McKeown, D. 1999. Experimental otter trawling on a sandy bottom ecosystem of the Grand Banks of Newfoundland: Analysis of trawl bycatch and effects on epifauna. Mar. Ecol. Prog. Ser., 181: 107–124.

Schwinghamer, P., Guignè, J.Y., and Siu, W.C. 1996. Quantifying the impact of trawling on benthic habitat structure using high resolution acoustics and chaos theory. Canadian Journal of Fisheries and Aquatic Sciences, 53: 288–296.

Steingrimsson, S.A. 2002. Potential coral reefs off the south coast of Iceland. Working paper to ICES Working Group on Ecosystem Effects of Fishing Activities meeting 2002.

Thrush, S., Hewitt, J., Cummings, V. J., and Dayton, P. 1995. The impact of habitat disturbance by scallop dredging on marine benthic communities: what can be predicted from the results of experiments? Mar. Ecol. Prog. Ser., 129: 141–150.

Tuck, I., Ball, B., and Schroeder, A. 1998. Comparison of undisturbed and disturbed areas. *In* IMPACT-II. The effects of different types of fisheries on the North Sea and Irish Sea benthic ecosystems. Ed. by H.J. Lindeboom and S.J. de Groot. NIOZ-Rapport 1998 – 1/RIVO-DLO Report CO 03/98.

Tuck, I.D. *et al*. 1999. The impact of water jet dredging for razor clams, *Ensis* spp., in shallow sandy subtidal environment. Journal of Sea Research, 43: 65–81.

Turner, S. J., Thrush, S., Hewitt, J. E., Cummings, V. J., and Funnell, G. A. 2000. Fishing impacts and the degradation or loss of habitat structure. Fish. Manag. Ecol., 6: 401–420.

Tyler-Walters, H., Hiscock, K., Lear, D.B., and Jackson, A. 2001. Identifying species and ecosystem sensitivities. Report to the Department for Environment, Food and Rural Affairs, from the Marine Life Information Network (MarLIN), Marine Biological Association of the UK, Plymouth, Contract CW0826 [Final Report].

Underwood, A. J. 2000. Trying to detect impacts in marine habitats: comparisons with suitable reference areas. *In* Statistics in ecotoxicology, pp. 279–308. Ed. by T. Sparks. John Wiley & Sons, Toronto.

Veale, L., Hill, A., Hawkins, S., and Brand, A. 2000. Effects of long-term physical disturbance by commercial scallop fishing on subtidal epifaunal assemblages and habitats. Mar. Biol., 137: 325–337.

Watling, L., and Norse, E. A. 1998. Disturbance of the sea bed by mobile fishing gear: a comparison to forest clearcutting. Conserv. Biol., 12: 1180–1197.

Witbraard, R., and Klein, R. 1994. Long-term trends on the effects on the southern North Sea beamtrawl fishery on the mollusc *Arctica islandica* L. (Mollusca, Bivalvia). ICES J. Mar. Sci., 51: 99–105.

10 MAINTENANCE OF GENETIC DIVERSITY AND APPROPRIATE FORMS OF MANAGEMENT

h) propose a process to be able to obtain information to develop advisory forms appropriate to the preservation of genetic diversity, beginning with the initiation of an evaluation of the advisory forms and management approaches that would be necessary and sufficient for the protection of genetic diversity of exploited stocks, and stocks suffering substantial mortality as by-catch.

10.1 Introduction

ToR h) has been included in response to a letter sent from the European Commission Director General, Fisheries to the General Secretary of ICES, on 25 September 2001, specifying that developing a framework for the preservation of genetic diversity is one of its “areas of immediate interest”. Preservation of biological diversity, including genetic diversity, also follows from the acceptance of the Rio Declaration, thereby creating a demand for developing management forms that can cope with this issue. Scientific justification for conserving genetic diversity within and among populations stems from several sources including: 1) maintaining adaptability of natural populations in face of environmental change; 2) future utility of genetic resources for medical and other purposes; and 3) changes in life history traits (e.g., age and size at maturation, growth) and behaviour (e.g., timing of spawning) that influence dynamics of fish populations, energy flows in the ecosystem, and ultimately, sustainable yield.

The impacts of fisheries on genetic diversity have received considerable attention in recent years in a wide variety of media, including journals, books, reports, conferences and workshops. Within ICES, the genetic effects of fishing have been included in the terms of reference for the ICES Working Group on the Application of Genetics in Fisheries and Mariculture (WGAGFM) in one form or another from 1995–2000, and were briefly discussed in the 2000 WGECO report.

Section 10.2 provides a background and assessment of the problems associated with the loss of genetic diversity through fisheries practices using Arctic cod as an example of loss of genetic diversity within a population. Section 10.3 suggests the most appropriate course of action to protect genetic diversity based on the best available scientific evidence, and Section 10.4 provides conclusions and recommendations for establishing a process for protecting the genetic diversity of exploited stocks and those suffering fishery-induced mortality.

10.2 Background

A gene is a hereditary unit that helps to determine a trait. The DNA sequence of a specific gene may not always be exactly the same. There may be some differences in the sequence, resulting in different variants of the same gene. Such alternate variants of a specific gene are called alleles and the number of different alleles is a measure of genetic variation. The different alleles of a specific gene often occur in different frequencies in different populations (allele or gene frequencies). The genetic variation of a species is therefore distributed both within populations, expressed as the different allele combinations between individuals (so-called genotypes), and between populations (in the form of differences in occurrence and frequency of alleles between populations). Each measure provides a different perspective on the genetic diversity of a population. Natural selection acts within populations, while the genetic potential of the species to adapt to environmental changes depends on the total genetic diversity represented among populations.

There are three general classes of threat to biodiversity at the gene level: 1) extinction (population or species), which results in complete and irreversible loss of genes; 2) hybridisation, which causes re-arrangement of co-adapted genes and loss of adaptability to local conditions; and 3) reduction in genetic variability within populations. This third threat can occur in a directed way (e.g., selective fishing) or due to a decrease in population size resulting in inbreeding (Laikre and Ryman, 1996).

Normally, marine fish have very large population sizes and the concern for loss of genetic diversity can appropriately be directed to loss of variation within populations through selection caused by fishing. In most marine species, the parents produce large numbers of offspring and there is large scope for local selection. However, when populations are very severely overfished to small numbers, concerns associated with small population size (e.g., number of actual breeders, inbreeding, etc.) and disruptions to migration between populations become prominent.

For a population, it is immaterial whether or not the mortality induced by fishing is incidental. Many by-catch and other non-target species are subject to substantial fisheries-induced mortality, given the vast areas of seabed trawled each year, and the unselective nature of most fisheries (Alverson *et al.*, 1994). Consequently, fisheries will also have genetic effects on non-target species.

The population structure of a species will determine the genetic impact of fisheries that results from the loss of spatial components. More subtle changes, inferred from phenotypic changes that are occurring irrespective of population abundance, may be more difficult to prove empirically, but can be estimated through modelling approaches. Consequently, objectives can be selected at a macro-level (e.g., number of spawning components, relative abundance of components, percent change in life history trait) to maintain genetic diversity under the precautionary approach. However, it will be more difficult to assign biologically meaningful reference points for these objectives. Unlike population dynamics models for which all parameters can be reasonably estimated and predictions can be made, we will never be able to predict what aspects of genetic diversity will be important for the future, nor what losses in the past have influenced present-day conditions.

10.2.1 Genetic variation among populations

Fishing is known to have an effect on the spatial structure of populations. The effect that this will have on genetic diversity will depend upon the migration patterns between these populations. New animals may migrate from one population to another, and if they mate within the new population, they have the potential of bringing new alleles to the local gene pool. This is called gene flow. There are many theoretical types of genetic population structure (cf. Smedbol *et al.*, 2002); these range from complete *panmixia* where each individual has an equal probability of reproducing with any other individual, to highly structured populations with complete reproductive isolation. Complete panmixia was postulated for the European eel, but this has since been disproved (Wirth and Bernatchez, 2000), and it is unlikely that panmixia occurs in marine species (although it is the null hypothesis for all genetic tests of population distinctness – see Section 10.2.2.1, below). At the other extreme, subdivided populations with reproductive isolation are also not typical, except in situations of rare and very localized species with limited possibilities for larval dispersal (cf. Nielsen and Kenchington, 2001; Smedbol *et al.*, 2002). While the genetic structure of marine species is generally not known, the *stepping-stone model* and its variants (Kimura and Weiss, 1964) are likely to be more relevant. In this model, a number of genetically distinct populations exist which are linked by gene flow. However, unlike Wright's (1931) *island model*, the probability of gene flow from one population to another is dependent on the geographic distance between populations. It is expected that genetic distance between populations will increase with geographic distance, i.e., there will be isolation by distance. A variant of this model is the source-sink model, where a stable population (source) contributes migrants to smaller populations (sinks) that only exist due to the recurrent contributions from the source population (cf. Smedbol *et al.*, 2002). It is critical to know the genetic structure of a species in order to infer the genetic implications of the loss of components. Unfortunately, there may be no outward appearances of population discontinuities. For populations linked by gene flow, the organization of these populations in time and space, along with the ratio of within- and among-population variation, is important to maintain in order to avoid negative genetic effects (Altukhov and Salmnekova, 1994). Fishing may result in the loss of populations, producing fragmentation and disruption of gene flow.

Taylor and Dizon (1999) describe the statistical approach commonly used to test for genetic structure among populations and discuss how this can result in management failure through loss of local populations. Typically, these tests are designed with the null hypothesis, H_o, being that populations are equal (panmictic), while the alternative hypothesis, H_A, is that populations are not panmictic. A standard $\alpha = 0.05$ is applied, placing emphasis on not concluding incorrectly that populations are genetically isolated, when, in fact, they are panmictic (a low Type I error). However, application of the precautionary approach might support the argument that it is a more serious error to incorrectly conclude that populations are panmictic when, in fact, they are reproductively isolated. In such cases, the statistical goal should be a low Type II error rate, even if this comes at the cost of a higher Type I error rate. These authors advocate calculating β, the probability of accepting the Null Hypothesis of panmixia when populations are actually isolated, as well as setting the more traditionally controlled α . Their intent is to avoid a "hidden" policy choice to treat one type of error as more serious than the other. In an example given in their paper, they illustrate how by choosing an $\alpha = 0.05$, a $\beta = 0.60$ is unintentionally accepted, giving a result that is 12 times (β/α) more likely to result in incorrectly pooling populations than an error that will incorrectly split them. In some cases, it might be appropriate to equalize these errors ($\alpha = \beta$), although this will inevitably require large sample sizes and/or an increased number of markers. As many genetic studies are undertaken without consideration of management questions, a careful evaluation of the methodology is needed to fully appreciate the applied implications of these studies.

10.2.2 Genetic variation within populations

Physical and life history traits (phenotype) are produced by the genetics of the individual, by the environment in which it lives (e.g., temperature, food availability), and by the interaction between the genes and the environment. Data on fish populations from many parts of the world have shown that removing large fish generally appears to favour the promulgation of slow-growing, early maturing fish (see reviews by Smith, 1999; Law, 2000). The challenge is to ascertain whether these changes are irreversible and a consequence of genetic alteration of the populations, or whether they are due to selected removals or a suite of other environmental factors such as temperature and prey fields. Put simply, is there a genetic difference between the fish removed and those left behind (Law, 2000)? Law and Grey (1989) and Heino (1998) have modelled the impact of a decline in age-at-maturation in Arctic cod, and conclusions of work in progress (U. Dieckmann, M. Heino and O.R. Godø) suggest that the phenotypic response is consistent with selection-induced deterioration of genetic diversity. However, empirical data for these conclusions are generally lacking in marine species, despite the fact that the evolution of life history traits is a field of great interest, both in population biology and genetics.

Modelling is a powerful tool both for exploring the expected consequences of current exploitation regimes, and for experimenting with different management measures that might be adopted to mitigate unwanted selection pressures. It can also be used to assess the scope of these problems, which can in turn be used in risk assessments. One of the areas in which we are data deficient is in the estimation of the proportion of phenotypic variance which is inherited. In terms of quantitative genetics, this proportion is referred to as the heritability of a trait (h^2), and traits with low values of h^2 change more slowly than those with higher values. Mean values of h^2 have been determined from broad surveys of both traits and species (Mousseau and Roff, 1987), and salmonids produce estimates consistent with these values (cf. Law, 2000). However, extrapolation from culture conditions to the wild can only be indicative, because the specific environment defines the heritability of a trait. Calculations of heritability from the wild are dependent on identifying kinship structure, an elusive property in most marine species due to the large population sizes. Roff (1997) suggests that, in the absence of better information, heritabilities for life history traits in the range 0.2–0.3 can be assumed, which means that 20–30 % of the observed variation is due to the genes, while the remaining 70–80 % is largely due to effects of the environment interacting with expression of those genes. To compensate for the lack of information on heritability, sensitivity analyses can be done using a range of heritabilities when modelling quantitative genetics and phenotypic data.

In the absence of direct genetic evidence, the dependence of phenotypes on environment can be characterized by a metric referred to as "reaction norms". The reaction norm predicts the phenotype that follows from a single genotype as a function of the condition of the environment. The reaction norms themselves are presumed to be genetically determined. Thus, change in a reaction norm is indicative of genetic change. The idea of using maturation reaction norms can be traced back to Stearns and Crandall (1984), Stearns and Koella (1986), and Rijnsdorp (1993). Probabilistic extension of the methodology is necessary to make the reaction norm approach fully operational (Heino *et al.*, 2002). Identification of traits under genetic selection using reaction norms may facilitate the identification of quantitative trait loci (QTLs) which could then be used to validate the models.

10.2.2.1 The special case of small populations

In all populations of a restricted size, the frequency of particular alleles changes randomly from one generation to the next. This process, called genetic drift, may also result in loss of genetic variation. By pure chance, some of the alleles that exist in the parent generation may not be passed on to their offspring. The smaller the population, the more dramatic the fluctuation of allele frequencies, and the faster the loss of genetic variation. Another consequence of small population size is inbreeding, i.e., the production of offspring from matings between close relatives. If a population is small and isolated, inbreeding is inevitable. In many species, inbreeding is coupled with reduced viability and reproduction, reduced mean values of meristic traits, as well as increased occurrences of diseases and defects, so-called inbreeding depression.

The rate of genetic drift and inbreeding is not determined by the actual (census) population size, N, but by a parameter denoted "effective population" size or N_e. Typically, estimates of N_e have large confidence intervals around them, especially when inferred from gene frequency data. In certain situations, N_e can be quite precisely estimated from abundance surveys, e.g., with the breeding population of the Atlantic right whale. Effective population size is nearly always less than N because generally not all individuals in a population are reproductive at spawning time. N_e depends on such factors as sex ratio, variance in family size (i.e., variability in numbers of offspring per individual), temporal fluctuations in numbers of breeding individuals, overlapping generations, etc. For example, for some species genetic variation will be reduced if the sex ratio of breeders departs from 1:1. It is much better genetically to have a population of 50 males and 50 females than to have one of 10 males and 90 females, yet both have 100 breeders. Similarly, the maximum genetic variation is produced in the population when all mating pairs produce equal-sized families. In the case of the northern elephant seal, dominant bulls establish a harem and monopolize females, skewing the sex ratio through mating behaviour (Hoelzel, 1999). Fishing practices that select one sex over the other also may, over time, cause a reduction of genetic diversity within populations.

Genetically small populations are unlikely to be of concern in marine fish with large census population sizes. For these species, commercial extinction is likely to occur long before populations are small enough to be inbred. However, hidden populations within management units may be fished to this level before the situation can be appreciated. Therefore, it is critical that the population structure of species be defined.

10.2.2.2 Case study of fisheries-induced selection on the northeast Arctic cod

The northeast Arctic cod (*Gadus morhua*) is one fish stock where consideration of genetic changes caused by fisheries-induced selection has attracted attention. This stock is very large, and even when stock abundance reached record-low levels in the 1980s, the spawning stock consisted of tens of millions of fish. This description holds even if substructure is considered (Mork *et al.*, 1985). Thus, in this example, loss of genetic diversity in northeast Arctic cod is considered in the context of fisheries-induced selection (cf. Law, 2000).

During the first quarter of the 20th century, intensive harvesting of Arctic cod took place on the spawning grounds which are at some distance from the feeding grounds. Under this scenario, cod with delayed maturation had a reduced mortality risk, while gaining in terms of increased size and, after maturation, increased fecundity. This historical selection pressure for delayed maturation may be responsible for the late maturation traditionally observed in this stock (Law and Grey, 1989); the median age-at-maturation was 10–11 years before the 1940s (Jørgensen, 1990). Since around 1930 when the modern trawler fishery started, harvesting became size-selective for larger fish, indirectly favouring selection for earlier maturation. Effort was also transferred to the feeding grounds. Borisov (1978) raised the concern that high fishing pressure might select for earlier maturation in this stock. Indeed, the decline in average age-at-maturation in this stock has been particularly strong (Jørgensen, 1990), and the year-classes born in the 1980s have a mean age-at-maturation of 6–7 years (Godø, 2000). Size-at-maturation has declined in parallel, from 89 cm (1940 year-class) to 74 cm (1989 year-class) (Godø, 2000). Assuming a cubic relationship between length and weight, this corresponds to a 42 % decrease in weight of the first-time spawning cod (assuming a constant fecundity-to-weight ratio, the same decrease applies to fertility).

Analysis of the reaction norms for age- and size-at-maturation for this stock shows a significant temporal trend towards higher probability of maturation at a certain age and size (M. Heino, U. Dieckmann and O. R. Godø, work in progress). A quantitative genetics model is currently being developed by this group to determine whether the observed rate of change is consistent with the selection pressures that have been present.

Although there may be environmental effects that are not considered in the reaction norm analysis, it is probable that the change in reaction norms of the northeast Arctic cod has a genetic basis. However, the analysis also shows that phenotypic plasticity (in form of the so-called "compensatory response", i.e., maturation at an earlier age correlated with a higher growth rate) also explains an important part of the observed changes in age- and size-at-maturation. Partitioning of response to genetic and phenotypically variable components is unfortunately not straightforward because of the interaction between these two factors discussed previously.

Theoretical studies indicate that decline in age-at-maturation could cause a major decline in sustainable yield from the northeast Arctic cod (Law and Grey, 1989; Heino, 1998). It must be emphasized that these models were designed to make only qualitative predictions and that the predictions on yield should be interpreted cautiously. Nevertheless, annual losses in sustainable yield of the order 10^5 tonnes appear to be possible. Thus, despite the uncertainty, these findings call for increased awareness of the possibility of adverse effects on yield. Earlier maturation will also result in smaller size-at-age after maturation and, assuming that large fish are more highly valued than small fish, diminish the market value of the catch. In addition, it is possible that earlier maturation may further increase recruitment variability in this stock. The long spawning migration imposes an energetic stress that would be relatively larger for smaller individuals, and may affect egg quality in females. If feeding conditions before the migration are poor, the energetic stress might become too high for the fish maturing at small size, and they might either fail to reach the spawning grounds or skip the spawning altogether. Likelihood of recruitment failure under poor conditions could therefore increase. On the positive side, it is unlikely that the stock could sustain the present-day exploitation regime if its maturation reaction norm was similar to its state prior to modern exploitation.

Management measures that would be necessary to mitigate selection pressures towards earlier maturation are, at the broad level, theoretically well understood in the case of northeast Arctic cod (Law and Grey, 1989; Heino, 1998). The origin of selection pressure is the shift of exploitation pattern: from selective removal of mature cod to unselective (with respect to maturity status) removal of both immature and mature cod. Increasing fishing pressure on mature fish and decreasing fishing of immature cod would diminish—and eventually revert—the selection on maturation given the large population size of the stock. However, the exact levels of selective and non-selective fishing mortality that would eliminate the selection pressure are not known, although the existing modelling results indicate that the emphasis should be strongly on selection for mature cod (Law and Grey, 1989; Heino, 1998). Size-selective harvesting strategies which allowed the escape of undersized fish could potentially prove to be an alternative way of mitigating selection pressures towards earlier maturation. This possibility remains currently unexplored, although the evaluation would be technically possible and practically feasible.

One further consideration is that selection pressures are not necessarily symmetric. Fishing can create a very strong selection gradient for early maturation, whereas in the absence of fishing, late maturity is only weakly selected for (Law and Grey, 1989; Rowell, 1993; Heino, 1998). Decreasing fishing pressure helps to decrease the selection pressures but may not easily reverse them. Thus, trying to restore genetic stock properties by reverting selection pressures is inherently more difficult than trying to slow down changes by decreasing the selection pressures. Thus, there is considerable uncertainty surrounding the management implications. However, under the precautionary approach to fisheries (FAO, 1996), “where there are threats of serious or irreversible damage, lack of full scientific complexity shall be not used as a reason for postponing cost-effective measures to prevent environmental degradation” (excerpt from the Principle 15 of the Rio Declaration of the UN Conference on Environment and Development, Rio de Janeiro, 1992). Therefore, there appears to be a strong case for incorporating a consideration of the genetic effects of fishing into the management of the northeast Arctic cod.

10.3 Managing genetic diversity

In this section, a scientific framework for the provision of advice on genetic diversity is outlined. We propose a three-phase approach to the development of this advice: identification of management objectives, identification of appropriate reference points and/or definition of acceptable risk, and development of a monitoring programme (Figure 10.3.1). Considerations for defining management objectives for maintaining genetic diversity within a species include:

1) genetic diversity among populations;

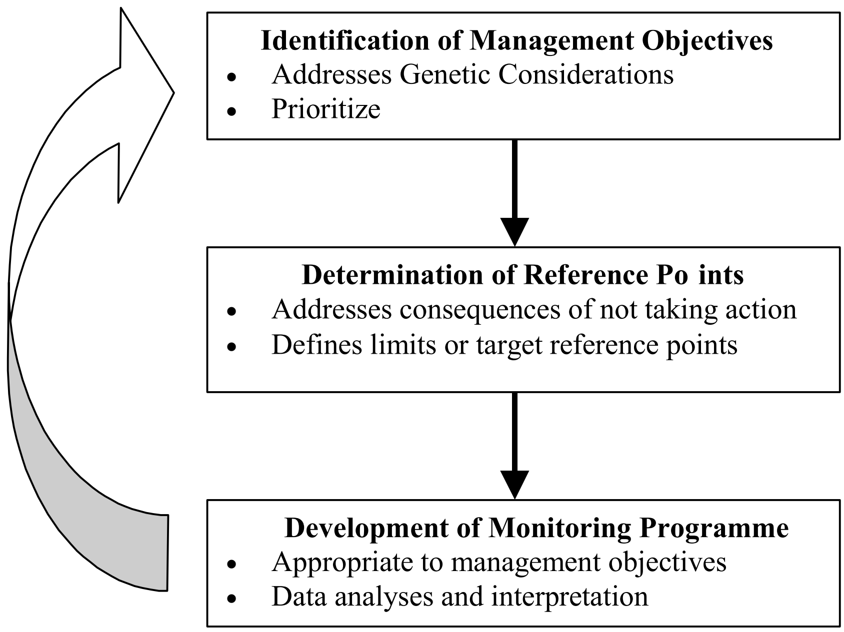
2) population structure and relative abundance;

3) within-population genetic diversity;

4) the current status of the species (endangered, threatened etc.).

The last consideration can be used to prioritize decision-making. This will be important because the management actions which are required when viable population sizes are intact are different from those needed when populations are small.

Figure 10.3.1. Three phases of approach to the development of advice for maintaining genetic diversity.



10.3.1 Management objectives

Any management regime requires clear management objectives that can be rendered operational. In examples from the literature, genetic diversity itself (e.g., number of alleles or genotypes) is not directly "managed" but the elements that influence it are. Thorpe *et al.* (1995) have suggested that the first priority should be to maintain populations in a natural setting to which adaptation may have occurred, and in which evolutionary forces may continue to act. Taylor and Dizon (1999) describe two similar objectives used by the U.S. Southwest Fisheries Science Centre in La Jolla, California: (i) maintain populations; and (ii) maintain the full geographic range of a species. Both of these examples address Consideration (1) and to a certain extent Consideration (3), however, they do not directly address loss of genetic diversity within populations due to selective fishing or the relative abundance of populations. The latter is important in maintaining migration patterns (gene flow) and population structure, both of which are potential consequences of exploitation. Examples of management objectives, which match those considerations, are provided in Table 10.3.1.1.

Table 10.3.1.1. Examples of management objectives to address generic concerns related to the loss of genetic diversity in marine species.

Consideration	Example Management Objective
Genetic diversity among populations	Maintain number of populations
Population structure and relative abundance	Maintain relative size of populations
Within-population genetic diversity	Maintain large abundance of individual populations
	Minimize fisheries-induced selection

With respect to genetic impoverishment caused by selection, the options can be broken down further: slow/stop/reverse fisheries-induced selection on X. It is necessary to specify which component of selection is being addressed ("X"), e.g., selection on maturation, sex, etc. Also, as discussed in Section 10.2.2.2, above, the management actions need to be specifically targeted if a reversal of selection pressure is desired, as opposed to slowing selection down. This may involve gear modification such as changes in mesh size, separator panels, or square mesh panels to alter selection and allow fish to escape. So that the "unpacked" objective becomes: Stop fisheries-induced selection on age-at-maturation.

10.3.2 Reference points?

The ICES framework for applying reference points to management objectives can also be applied to genetic diversity objectives. However, while target reference points may be established, reference points and limit reference points, as defined by ICES, are more problematic. ICES defines reference points as "specific values of measurable properties of systems (biological, social, or economic) used as benchmarks for management and scientific advice" (ICES, 2001). Their purpose is to flag decision points and, therefore, the consequences of not taking an action at a particular reference point should be clear.

One of the difficulties with determining minimum acceptable levels of genetic diversity is that we do not know precisely what aspect of genetic variability will be important for a species to adapt to environmental change in the future. We can deduce that genes under selection, that is quantitative trait loci, will be important; however, very few of these have been identified for any species. When phenotypic traits are used as a proxy of genetic diversity, it is easier to quantify the outcome of following the management advice. Here, modelling has an important role in predicting the consequences of decisions and, in particular, models that incorporate population and quantitative genetics are powerful. However, we do know that specific actions will lead to a negative effect, and these can be avoided. For example, we know that in most cases loss of populations will result in loss of genetic diversity, although we cannot say that losing 1 of 5 is acceptable but losing 2 is not. Target reference points are "properties of stocks/species/ecosystems which are considered to be desirable from the combined perspective of biological, social, and economic considerations" (ICES, 2001). For genetic diversity, target reference points can be established. The biological target would be no loss, modified by social and economic considerations (Table 10.3.2.1).

Table 10.3.2.1. Example biological target reference points for proposed management objectives with an example of a limit reference point (others to be determined (TBD)).

Proposed Management Objective	Example Target Reference Point (Biological Perspective)	Example Limit Reference Point
Maintain number of populations	Maintain all populations	TBD
Maintain relative size of populations	Maintain relative size of populations within X % of each other	TBD
Maintain large abundance of individual populations	Maintain abundance of individual spawning population above X %	$N_e >> 5{,}000$ spawners
Minimize fisheries-induced selection	No fisheries-induced selection	TBD

Limit reference points are "a value of a property of a resource that, if violated, is taken as *prima facie* evidence of a conservation concern. By 'conservation concern', ICES means that there is unacceptable risk of serious or irreversible harm to the resource…" (ICES, 2001). Loss of *alleles* from a species represents an *irreplaceable* component of genetic diversity. The *irrevocability* of genetic loss, combined with our inability to assess the consequences of not taking action, result in greater potential risks associated with any decision-making process that allows for loss of diversity. Loss of alleles may qualify as a conservation concern if the risk is judged unacceptable, however determining the limits at which the resource is "harmed" will be problematic for the reasons discussed above. In this case, the limit reference point may be very high and close to the target reference point.

Because changes in *allele frequency* may be irreversible or at best very difficult to reverse, limit reference points are likely to have to be set very conservatively because the negative consequences of exceeding the limit reference point will be difficult, if not impossible, to subsequently rectify. Nevertheless, limit reference points could be defined for some objectives, especially those applicable to within-population genetic diversity (Table 10.3.2.1). For example, recent theoretical work suggests that successful breeding population sizes of 1,000 to 5,000 are required for long-term population viability (Lynch and Lande, 1998). If limit and/or target reference points can be established, genetic risk assessment (e.g., Currens and Busack, 1995; Allendorf *et al.*, 1987) may provide a framework for decision making in light of uncertainty and consideration of other factors (e.g., biological, economic and social).

10.3.3 Monitoring genetic changes

Methods selected for monitoring genetic diversity will depend upon the management objective. An effective monitoring programme requires three phases: identifying monitoring questions, identifying monitoring methods, and the analysis and interpretation of information for integration into management strategies and the refinement of management objectives

(Gaines *et al.*, 1999). Examples of monitoring questions include: What is the genetic diversity within a population or among populations? How has habitat fragmentation affected the genetic structure of a population or species (cf. Gaines *et al.*, 1999)?

Once the questions are established, the monitoring methodology can be determined. This includes both sampling design and choice of markers, as well as consideration of derived indices. Genetic diversity can be measured at many different levels using a variety of markers. Markers that are ideal for identifying population structure (e.g., so-called neutral markers such as nuclear microsatellite arrays) are not useful for monitoring traits under selection. Different types of markers or combinations of markers can be used to monitor temporal changes in genetic diversity to address specific questions related to the management objectives. With the development of high-throughput equipment with low operating costs, genetic monitoring programmes have become affordable. However, an important constraint on addressing monitoring questions is the lack of historical data. Even where tissue exists, it is often preserved in formaldehyde, rendering the extraction of good quality DNA potentially difficult. Given this constraint, it is recommended that tissue samples from research vessel survey catches be archived for future genetic analysis. The amount of tissue needed for genetic work is very small and hair, scales and otoliths can be used.

In monitoring phenotypic traits, existing biological data from fisheries surveys is generally adequate to identify potential cases where fishing may have caused selection. However, it is important to take direct environmental effects into account in order to disentangle the genetic component of variation. This requires either monitoring quantities that are robust to environmental variations, or monitoring, in addition to phenotypic traits, the relevant environmental variables that have a major influence on the phenotypic traits in consideration. The former option is preferable when possible. Reaction norms are an example of quantities that are robust to environmental variations. In particular, reaction norms for age- and size-at-maturation are expected to be useful for monitoring changes in maturation.

10.4 Conclusions and procedural recommendations

It is clear from the above discussion that it would be possible to develop advisory forms appropriate to the preservation of genetic diversity. While this ToR begins the process for obtaining the information to be used on these forms, the process for advice should involve collaboration between WGECO and the Working Group on the Application of Genetics in Fisheries and Mariculture (WGAGFM); the latter body has already contributed to the discussion of this topic. To this end, WGAGFM has agreed to propose a ToR for consideration in 2003: "*To develop practical management options for the conservation of genetic diversity in marine fish and shellfish of economic importance*". The response to this ToR can then be shared with WGECO according to the schedule outlined in Table 10.4.1, to recommend to ACE changes to the advisory forms in 2004 to address concerns over loss of genetic diversity expressed by the EC DG Fisheries.

Table 10.4.1. The proposed process for the continued development of a mechanism for ICES to provide for "genetic diversity" in management advice.

Task	Lead party(ies)	Example Timeframe
1. Review and development of considerations for maintaining genetic diversity	WGAGFM	March 2003
2. Review and development of management objectives to address genetic considerations	WGAGFM; WGECO	spring 2003
3. Evaluation of reference points and/or consequences of not addressing management objectives	WGECO; WGAGFM	spring 2003
4. Development of a list of quantifiable variables whose values, individually or in combination, identify a significant threat to genetic diversity	WGECO; WGAGFM	spring 2004
6. Case studies reviewed under the proposed framework and strengths and weaknesses determined	WGAGFM; WGECO	spring 2004
7. Assessment of possible management responses for protection of genetic diversity and provision of commentary to ACE	WGECO; WGAGFM	spring 2004
8. ACE formulates advice to ICES customers	ACE	September 2004

10.5 References

Allendorf, F., Ryman, N., and Utter, F. 1987. Genetics and fishery management: Past, present, and future. *In* Population Genetics and Fishery Management, pp. 1–19. Ed. by N. Ryman and F. Utter. Washington Sea Grant Publications, University of Washington Press, Seattle and London.

Altukhov, P., and Salmnekova, E.A. 1994. Straying intensity and genetic differentiation in salmon populations. Aquaculture and Fisheries Management, 25 Supplement, 2: 99–102.

Alverson, D.L., Freeberg, M.H., Murawski, S.A., and Pope, J.G. 1994. A global assessment of fisheries bycatch and discards. FAO Fisheries Technical Paper 339, FAO, Rome.

Benton, T.G., and Grant, A. 1999. Elasticity analysis as an important tool in evolutionary and population ecology. Trends in Ecology and Evolution, 14: 467–471.

Borisov, V.M. 1978. The selective effect of fishing on the population structure of species with long life cycle. Journal of Ichthyology, 18: 896–904.

Currens, K.P., and Busack, C.A. 1995. A framework for assessing genetic vulnerability. Fisheries, 20: 24–31.

FAO. 1996. Precautionary approach to capture fisheries and species introductions. Elaborated by the Technical Consultation on the Precautionary Approach to Capture Fisheries (Including Species Introductions). Lysekil, Sweden, 6–13 June 1995. FAO Technical Guidelines for Responsible Fisheries 2. FAO, Rome.

Gaines, W.L., Harrod, R.J., Richy, J., and Lehmkuhl, J.F. 1999. Monitoring biodiversity: quantification and interpretation. Gen. Tech Rep. PNW-GTR-443. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, Oregon. 27 pp.

Godø, O.R. 2000. Maturation dynamics of Arcto-Norwegian cod. IIASA Interim Report IR-00-024. International Institute for Applied Systems Analysis, Laxenburg, Austria.

Heino, M. 1998. Management of evolving fish stocks. Canadian Journal of Fisheries and Aquatic Science, 55: 1971–1982.

Heino, M., Dieckmann, U., and Godø, O.R. 2002. Measuring probabilistic reaction norms for age and size at maturation. Evolution, in press.

Hoelzel, A.R. 1999. Impact of population bottlenecks on genetic variation and the importance of life-history; a case study of the northern elephant seal. Biological Journal of the Linnaen Society, 68: 23–39.

ICES. 2001. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 2001/ACME:09.

Jørgensen, T. 1990. Long-term changes in age at sexual maturity of northeast Arctic cod (*Gadus morhua* L.). J. Cons. Cons. Int. Explor. Mer., 46: 235–248.

Kimura, M., and Weiss, G.H. 1964. The stepping-stone model of population structure and the decrease of genetic correlation with distance. Genetics, 49: 561–576.

Laikre, L., and Ryman, N. 1996. Effects on intraspecific biodiversity from harvesting and enhancing natural populations. Ambio, 25: 504–509.

Law, R. 2000. Fishing, selection, and phenotypic evolution. ICES Journal of Marine Science, 57: 659-668.

Law, R., and Grey, D.R. 1989. Evolution of yields from populations with age-specific cropping. Evolution and Ecology, 3: 343–359.

Lynch, M., and Lande, R. 1998. The critical effective size for a genetically secure population. Animal Conservation, 1: 70–72.

Mork, J., Ryman, N., Ståhl, G., Utter, F., and Sundnes, G. 1985. Genetic variation in Atlantic cod (*Gadus morhua*) throughout its range. Canadian Journal of Fisheries and Aquatic Science, 42: 1580–1587.

Mousseau, T.A., and Roff, D.A. 1987. Natural selection and the heritability of fitness components. Heredity, 59: 181–197.

Nielsen, E.E., and Kenchington, E. 2001. Prioritising marine fish and shellfish populations for conservation: A useful concept? Fish and Fisheries, 2: 328–343.

Rijnsdorp, A.D. 1993. Fisheries as a large-scale experiment on life-history evolution: disentangling phenotypic and genetic effects in changes in maturation and reproduction of North Sea plaice, *Pleuronectes platessa* L. Oecologia, 96: 391–401.

Roff, D. A. 1997. Evolutionary Quantitative Genetics. Chapman and Hall, New York.

Rowell, C.A. 1993. The effects of fishing on the timing of maturity in North Sea cod (*Gadus morhua* L.). *In* The exploitation of evolving resources, pp. 44–61. Ed. by T.K. Stokes, J.M. McGlade, and R. Law. Springer-Verlag, Berlin.

Smedbol, R.K., McPherson, A.A., Kenchington, E., and Hansen, M.M. 2002. Metapopulations in the marine fish literature: the use and misuse. Fish and Fisheries (in press).

Smith, P.J. 1999. Genetic resources and fisheries: policy aspects. *In* Towards Policies for Conservation and Sustainable Use of Aquatic Resources. Ed. by R.S.V. Pullin, D.M. Bartley and J. Kooiman. ICLARM Conference Proceedings, 59: 43–62.

Stearns, S.C., and Crandall, R.E. 1984. Plasticity for age and size at sexual maturity: a life-history response to unavoidable stress. *In* Fish Reproduction, pp. 13–33. Ed. by G. Potts and R.J. Wootton. Academic Press, London.

Stearns, S.C., and Koella, J.C. 1986. The evolution of phenotypic plasticity in life-history traits: predictions of reaction norms for age and size at maturity. Evolution, 40: 893–913.

Stephenson, R.L., and Kenchington, E. 2000. Conserving fish stock structure is a critical aspect of preserving biodiversity. ICES CM 2000/Mini:07.

Taylor, B.L., and Dizon, A.E. 1999. First policy then science: why a management unit based solely on genetic criteria cannot work. Molecular Ecology, 8 (Suppl. 1): S11–16.

Thorpe, J.E., Gall, J., Lannan, J.E., Nash, C., and Ballachey, B. 1995. The conservation of aquatic resources through management of genetic risks. *In* Conservation of Fish and Shellfish Resources: Managing Diversity, pp. 33–46. Ed. by J. Thorpe, G. Lannan, and C. Nash. Academic Press, San Diego.

Wirth, T., and Bernatchez, L. 2001. Genetic evidence against panmixia in the European eel. Nature, 409: 1037–1040.

Wright, S. 1931. Evolution in Mendelian populations. Genetics, 16: 97–159.

11 ECOLOGICAL DEPENDENCE: HOW CAN THIS BE INCORPORATED INTO MANAGEMENT ADVICE?

11.1 Introduction

This section addresses our Term of Reference i)… "propose a process to be able to obtain information to consider 'ecological dependence in management advice, firstly addressing the groups of species with the ecological linkages that are known with high reliability to have strong ecological linkages', including specification of the data requirements and models that would be required to provide the scientific basis for a response to that request. Propose a workplan and timetable for ICES to prepare itself for developing that scientific advice".

This work is required to underpin the provision of advice to the European Commission. The understanding and provision of protection for species that are ecologically dependent on other species affected by fisheries (i.e., those with strong ecological linkages) is one of the three most immediate areas where management advice needs to adopt a wider "ecosystem" approach (ICES, 2001). The other two areas are "impacts on non-target species and sensitive habitats" (see Section 9) and "preservation of genetic diversity" (see Section 10). It must also be recognised that the high priority should be accorded to protection of habitats which are essential to or are themselves "species at risk" (equivalent to "threatened or declining" in some contexts).

In developing our thinking on this issue we have taken ecological dependence to mean a need of a species for a particular aspect of the habitat (physical, chemical or biological) or through ecological linkages within the food web. This includes both vertical links – species and their predators and species and their resources, and horizontal interactions such as competition for food or space.

WGECO deconstructed this task into six tasks:

1) the characterisation of ecological linkages;
2) definition of what constituted a strong linkage;
3) presenting illustrations of what we know, or can infer, about such linkages within the ICES area;
4) presentation of guidance to Working Groups and Advisory Committees for how to identify the presence and importance of strong ecological linkages in case-by-case applications;
5) illustrations of the spectrum of how ecological linkages should be taken account of in scientific advice;
6) a workplan for developing advice about the issues highlighted in 3), 4), and 5).

We initially distinguish two classes of ecological linkage:

1. biotic;
2. habitat.

For each of these, we describe the biological basis of the linkage and provide some illustrative examples. We then consider criteria that might be used to characterise an example of a strong linkage of each type. In doing this we consider the data and modelling required in order to make such an assessment.

11.2 Biotic linkage

11.2.1 Biological basis for linkages

The primary linkages will be through biotic interactions. These may lead to unwanted indirect effects of exploiting one species on others through ecological dependency. Biological interactions may take different forms, but they are generally trophic in origin. Ecosystem components are linked through trophic relationships, either "vertical" through predator-prey interactions or "horizontal" through competition in exploiting a common food resource. Parasite-host interactions represent a specific type of complex predator-prey relationship involving sometimes different host species and often a particular epidemiology. Other, non-trophic, relationships involve for instance competition for space and commensalisms. Spatial competition may occur frequently during specific life stages in the recruitment process. For instance, spatfall of molluscs may be enhanced by opening up empty space. Commensalism, and other intricate biological interactions, are not well described for temperate areas, but do occur (e.g., the fish *Echiodon drummondi* inhabiting the intestine of sea cucumbers; and the hermit crab *Pagurus prideauxi* associated with the cloak anemone *Adamsia carciniopadosi*).

A derivative of these biotic linkages is that of technical interactions. Technical interactions cause the familiar by-catch and discard problem: the ecological constraints on individual species operate in such a way that they share a common habitat and therefore exploiting one species leads to an unavoidable by-catch of others (a forage fish fishery may result in a by-catch of predators; the gadoids caught in demersal fisheries may largely compete for the same food resources). Although this problem may often be reduced by prescribing gear specifications that take behavioural differences into account, in practice there are limitations to their implementation, particularly if the economics of a particular fleet are based on the value of a multispecies catch.

Finally, ecological dependence in management advice may apply to individual species performing part of their life history in different management areas. Salmon and eel are extreme cases in this respect, but many top predators of concern (e.g., various sharks) have unit stocks that incorporate two or more fisheries management areas. Any advice on these should take account of all factors affecting them over the entire distribution area of the stock.

11.2.2 Assessing ecological dependence

In order to take ecological dependence into account when formulating management advice, the following stages can be distinguished:

- *Identification of existing and potential links*: For many species, information exists on their food preference or habitat requirements and in effect this describes their realized niche. To what extent a particular component is affected by changes in another component to which it is linked depends on the potential of expanding its niche by switching to other food sources or habitats. If there is insufficient information on feeding preferences or habitat requirements to determine this potential, it may be derived from other characteristics of the species such as its morphological limitations to utilize a specific food source or habitat or by analogy to related species.
- *Determine the strength of these links*: In the case of parasite-host and commensal relationships, linkages often obligatorily involve individual species (sometimes higher taxa) and therefore must be considered "strong". However, predator-prey relationships among marine organisms appear to be, within broader taxonomic prey categories, rather flexible and few if any predators depend obligatorily on a single prey type. Nevertheless, it seems also clear that when species are taken together in larger units, as for instance is commonly done in ecosystem models, the strength of the linkages increases. In a management context, ecological dependence does not necessarily refer to interactions between individual species, but could also refer to larger groupings (e.g., forage fish, epifauna consumers). In this case, we need a pragmatic approach to distinguish between strong and weak linkages. As a general rule, we might consider a linkage strong if a change in the dynamics of one species ALWAYS resulted in a measurable change in the dynamics of the other. This ability will be constrained by the analytical and sampling procedures having sufficient power. For example, from a predator point of view, a linkage with a prey might be called strong if the predator would not be able to replace the average contribution of that prey to the diet, if its abundance was significantly reduced. This situation appears to be for instance the case with common guillemot and Arcto-Norwegian cod in relation to capelin (Vader *et al.*, 1990; Nakken, 1994). It might be possible to develop empirical rules for this along the lines that prey categories representing more than x % of the total diet at any particular life stage could serve as a criterion for (potentially) strong linkages. From a prey point of view, a linkage might be called strong if a predator (group) accounts for a relevant proportion of the total natural mortality rate. Again, we might take a value, y %, as a criterion, because it would seem unlikely that other predators might replace this predation mortality completely if that predator became extinct. It is noteworthy that these definitions of strong linkages in predator-prey relationships may not be symmetrical: the linkage may be strong in terms of prey mortality and weak in terms of diet fraction of the predator or the other way round! The strength of a link (e.g., proportion of the predator's diet or prey's mortality) can be assessed using foodweb models that estimate the flow of energy through the trophic network. However, where data are available that allow a more direct quantification of the link (e.g., stomach analysis data), the use of these data is preferred.
- *Relationship with management*: Ecological dependencies represent the second-order effects of human activities that should be taken into account when managing first-order effects. For instance, river runoff of nutrients may have to be managed on the basis of the effects of coastal eutrophication on productivity and of the risk of anoxia. In the case of fisheries, exploiting a predator releases predation mortality on its main prey, which translates into a positive population trend and therefore might be of less interest from a conservation and management point of view. In contrast, exploiting an important forage fish resource may lead to a reduction in total consumption by dependent predators and translate into negative population trends of their populations. The issue of an ecosystem approach to marine management would therefore appear to apply particularly to bottom-up rather than top-down processes and linkages. The strong linkages within ecosystems that have been identified so far and that might have to be taken into account explicitly in management advice may be grouped among the following headings: (1) dependence of productivity on nutrients; (2) dependence of predators on prey; and (3) dependence of scavengers on discards and offal. The provision of appropriate management advice, taking into consideration strong linkages among ecosystem components, depends to a large extent on the type of ecological dependence and on the type of management problem envisaged. These two issues also determine the kind of data as well as models required. For instance, predator-prey relationships among target fish species are important in assessing the effect of, for instance, changes in mesh sizes on future yields. This problem has been addressed by the Multispecies Assessment Working Group by collecting comprehensive diet data sets as snapshots for a limited number of years and applying Multispecies Virtual Population Analysis. Undoubtedly, other strong linkages through predator-prey relationships between non-target species and commercial fish species would similarly require reliable diet data, but the models to be developed might have to be coarser because of lack of detailed information on the population structure of these stocks. For discards and offal, experiments determining the fate of these by-products appear to be more suitable, because random sampling of, for instance, bird diet data is problematic. Thus, it appears that there can be no general guidelines as to what data and model requirements can be linked to management issues related to ecological dependence. Rather, once specific issues have arisen, the next step will be to determine how the problem can be addressed.

11.2.3 Examples

11.2.3.1 Sandeels and predators

In 1999, ICES was asked by the EC to advise on management measures for the industrial fisheries for sandeels, taking into account any dependence of seabirds on sandeels as prey. The ICES Study Group on Effects of Sandeel Fishing (SGSEF) first reviewed the role of sandeels in the diets of seabirds around the North Sea. It concluded that there was a strong ecological linkage between some species of seabirds as predators and sandeels as prey. The linkage was particularly strong during the breeding season when the foraging range of seabirds was limited by travel time from breeding sites. In those cases, breeding failures of some species were clearly associated with local shortages of sandeels, although breeding could fail for reasons other than shortage of sandeels, and the strength of the dependence varied among species of seabirds. The combination of a relatively high ecological dependence and an effective monitoring programme of breeding success, led the Study Group to conclude that kittiwake breeding success was a particularly good indicator of sandeel availability to coastal seabirds during the breeding season. After reviewing all the information, the SGSEF recommended, and ICES advised, that a decision rule be implemented in management of the sandeel fishery, to account for this ecological dependence. The rule states that when kittiwake breeding success falls below 0.5 chicks per well-built nest for three consecutive years, all sandeel harvesting within 50 km of the UK coast should be stopped (ICES, 2000a).

11.2.3.2 Baltic three-species model

Gislason (1999) developed a model which includes inter-specific effects for cod, herring, and sprat in the Baltic Sea. It includes both top-down (predation mortality) and bottom-up (ration-mediated growth). The Baltic fish community is relatively simple, dominated by cod, herring, and sprat. Ecological linkages are stronger when the system is dominated by a few species. Cod are cannibalistic and eat both herring and sprat. Both herring and sprat eat 0-group cod. For status quo SSB, the magnitude of the effects of including ecological interactions can be assessed from Table 11.2.3.2.1.

Table 11.2.3.2.1. Status quo SSB (kt) and the change relative to the single-species estimates are in brackets from Gislason (1999).

	Cod	Herring	Sprat
Single species	221	970	628
Predation	233 (5 %)	1610 (66 %)	939 (50 %)
Predation and Ration	330 (49 %)	1510 (56 %)	826 (32 %)

The models used to estimate the interactions were the standard population (VPA), fish physiology models, and predation models. The data requirements for the population reconstruction are age-structured survey and catch data. The physiological models require data on energetics, growth, and maturity. The interactions are defined from fish stomach analyses. Knowledge of the rest of the ecosystem is also required which is based mainly on surveys.

Gislason concludes that, when biological interactions are taken into account, reference limits for forage fish cannot be defined without consideration of the biomass of their predators. Similarly, reference points for the predators must include the biomass of their prey.

11.3 Biota-habitat linkages

11.3.1 Biological basis for ecological linkage to habitat

The physical nature of the sea floor is an important determinant of benthic community composition (Rhoads and Young, 1970; Gray, 1981; Rhoads and Boyer, 1983). Sediment parameters tend to correlate with depth and the hydrographic regime (current flow, wave action). Later works have expanded this framework to consider the role of organic matter flux to the sediment (Pearson and Rosenberg, 1978, 1986). Some of the clearest examples of the importance of the physical environment for benthic organisms arise from anthropogenic alteration of the habitat which leads to an altered biological assemblage; this holds whether the impact is caused by dumping (Herrando-Perez and Frid, 1998) or by fisheries (Lindeboom and de Groot, 1998; Tuck *et al.*, 1998).

Bottom fishing gears impact on the sea floor, causing mortality or injury to surface-living and shallowly buried fauna (Tuck *et al.*, 1998), altering physical habitat features (Auster *et al.*, 1995), sedimentation (Churchill, 1989) and nutrient cycling (Mayer *et al.*, 1991). In the short term, such disturbed areas may attract mobile scavengers and predators (Kaiser and Spencer, 1994). It is difficult to assess the impact of these changes at the scale of the ecosystem (Thrush *et al.*, 1998) and the lengthy time scales over which exploitation has occurred (Frid *et al.*, 2000).

The changes in the physical habitats brought about by dynamite fishing are a particularly clear example of fishing-induced habitat modification. However, all mobile bottom gears impose some degree of alteration on the habitat. This might range from minor short-term changes due to the resuspension of sediments, through alterations in particle size (by winnowing the finer sediments away, or removal of boulders) to loss of biogenic structures such as tube worm or sea-fan beds or *Lophelia* reefs (Section 7). Many nearshore and shallow-water environments are naturally exposed to physical disturbance from storm waves and tidal currents, and the biota in these areas are adapted to cope with these stresses.

The scale of these impacts varies with fishing gear type (scallop dredges have more impact than beam trawls, beam trawls with heavy chains will have more impact than otter trawls) (ICES, 2000b). Habitats also vary in their vulnerability (Section 9). A sandy sediment with no structural fauna may show no effects of the trawl within days (possibly hours) of being impacted. In contrast, a single pass of a fishing gear through a bed of tube worms or erect sponges may change the habitat to one of unconsolidated sediment, which may then prevent colonisation by the original mix of taxa and so persist as an alternative stable state. These impacts are relatively easy to mitigate against: sensitive environments can only be effectively conserved in areas closed to fishing, while in other areas impacts can be reduced by alterations to the configuration of the fishing gear.

11.3.2 Assessing interaction strength with the habitat

Interaction strength or ecological dependence on habitat form is clearly a continuum. The most obvious examples of a strong interaction between a species and its habitat are those species that obligate on another, structure-building, species. Next we might consider species dependent on a single type of habitat characteristic or a narrow range of features (for examples, see Snelgrove and Butman, 1994). These might be considered, for the purposes of management, as strong, known with high reliability, linkages. The continuum extends through species with preferences for specific particle sizes or other habitat features but not exclusively these.

For some species, the ecological requirement for a particular habitat feature may be only seasonal. While this needs to be recognised, the compromising of a seasonally obligate need will have the same consequences for the organism as the effect of its loss on a species with a non-seasonal dependence.

From this it follows that obligate requirements by a species for specific habitat (physical, chemical or biotic) features should be regarded as a **Very Strong Ecological Dependence**. The lack of any other clear "break-point" on the continuum prevents the nomination of any additional categories of habitat linkage strength. However, it may be possible, on a case-by-case basis, to develop criteria along the lines of z % time spent in habitat *I* represents a strong dependence on that habitat. Obvious candidates in this next tier of considerations are spawning grounds, nursery areas, and feeding grounds.

The U.S. Congress defined essential fish habitat for federally managed fish species as "those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity." The conservation of essential fish habitat is an important component of building and maintaining sustainable fisheries (National Marine Fisheries Service Website 2002). Although the designations of essential fish habitat are directed to be based on "best available science", the designations are produced by a "public process with opportunities for input at many steps" (Section 303 – Magnuson-Stevenson Act).

In acting on this legislation, the National Marine Fisheries Service "has taken a broad view of habitat as the area used by fish throughout their life cycle. Fish use habitat for spawning, feeding, nursery, migration, and shelter, but most habitats provide only a subset of these functions. Fish may change habitats with changes in life history stage, seasonal and geographic distributions, abundance, and interactions with other species. The type of habitat, as well as its attributes and functions, are important for sustaining the production of managed species" (NMFS Source Document: Essential Fish Habitat: Life History and Habitat Characteristics. NMFS Website).

The broad interpretation of the Act was supported by the scientific community and at stakeholder consultations. Implementing it proved challenging and costly, however. The determination of essential fish habitat for the large numbers of marine and anadromous species has required significant time and money for the science and the consultation steps. Moreover, the inclusive definition of "essential fish habitat" has required that the designated habitats have been equally inclusive (31 examples available on Northeast Regional Fisheries Laboratory site on the NMFS site). The large amount of the sea and seabed included as "essential fish habitat" has made management decisions based on essential fish habitat impact on many human activities. These impacts, in turn, have prompted a large number of legal challenges to these designations. The challenges were to the process and not the science, but designations of essential fish habitat in the ICES area will require some bureaucratic process that will entrain substantial scientific effort.

Recent unfavourable court decisions on designations of critical habitat for terrestrial animals have led to the withdrawal of many designations of critical fish habitat under the National Environmental Protection Act, while the entire evaluation and designation process is reviewed (NOAA Constituent Release, March 3, 2002). The processes for designating "critical fish habitat" and "essential fish habitat" are not the same, but the demands for scientific support are high in both processes.

There are important lessons to be taken from the U.S. experience. Although there is sound scientific reason to define "essential habitat" broadly, particularly within a precautionary approach, there are costs to such an approach. A broad definition, addressing all life history stages and behaviours, will demand a great deal of scientific time, effort, and money to implement. Management measures based on broad definitions might also be perceived as intrusive by many stakeholders, and implementation might be resisted. On the other hand, of course, a very narrow definition of essential fish habitat might be less easy to implement, if substantial knowledge and site-specific data were required to determine what areas met the narrow definition. It might also not achieve the goal of protecting habitat needed for the species, if the definition was so narrow that not all the habitat needed for a rebuilt population were available. Finding a feasible scale on which to approach habitat designation and management is an important scientific task to be confronted early in any initiative on fish habitats.

Under the proposed definition of a Very Strong Ecological Dependence, the required assessment is simply one of obligate or not and the data requirements are straightforward arising from an understanding of the biology/ecology of the species. There is no obvious need to invoke ecosystem models to resolve this. Should other criteria be advanced, for example involving proportional use of a habitat, then spatially resolved population models including structured populations would be needed.

11.3.3 Examples

11.3.3.1 Herring and gravel beds

Herring spawn exclusively on gravel beds and the persistence of specific races or sub-populations depend on the continued presence of gravel at their traditional spawning locations (Maravelias, 1997). Clearly, licences for marine extraction should not be issued for gravel beds presently or formerly used as spawning grounds by herring. While trawling may locally cause surface disturbance of these beds by moving material around, a greater potential for harm to the habitat may be caused by deposits and smothering stemming from activities upstream of these beds. Immediate threats might be largely restricted to the spawning season, but persistent sedimentation may ultimately lead to spawning ground degradation and therefore require mitigating actions such as spatial and/or temporal closures.

11.3.3.2 *Lophelia* and associated species

The scleratinian reef-building coral, *Lophelia*, provides structural habitat to a wide variety of species, and diversity is approximately three times higher on the reefs compared to the surrounding soft bottoms (UKBAP, 2000). Unlike the case for tropical reefs, there are no known examples of species that are obligate reef dwellers. However, *Munidopsis serricornis*, *Ophiacantha* spp. and *Eunice* spp. all exhibit high abundance on the reefs and are seldom found in other Norwegian habitats (Fosså and Mortensen, 1998).

Recent observations around *Lophelia* reefs off Norway indicate that this habitat is preferred over other surrounding waters by both Norway redfish *Sebastes viviparus* and tusk *Brosme brosme* (Jensen and Fredericksen, 1992; Fosså *et al.*, in press). Both are believed to occur more generally over rocky or rough seabeds (Froese and Pauly, 2002), but evidence from both video surveys (Fosså *et al.*, in press) and fishermen indicates a dramatic fall in numbers of these fish if the *Lophelia* reef is destroyed. While it is plain that this is not an obligate association (as with herring and gravel beds (Section 11.3.3.1), trawling over *Lophelia* reefs seems likely to affect the populations of these species beyond the direct of capture in trawls. Neither fish species is subject yet to formal assessment and advice by ICES, but there is a commitment by the North Sea ministers to establish reference points for tusk by 2007. When such reference points are established, advice to avoid using towed gear in areas of *Lophelia* could be included.

11.4 Delivering scientific advice taking account of ecological dependence

11.4.1 Possible management responses

In 1999, WGECO (ICES, 2000b) considered the possible management responses to a number of ecosystem-level impacts of fishing activities. The exact management response that is required for a particular example of ecological dependence will need to be evaluated on a case-by-case basis. However, for strong ecological habitat linkages, the most effective control is likely to involve spatial closures possibly combined with gear exclusions and gear modifications. For strong ecological biotic linkages, spatial management measures are sometimes effective, whereas in other cases they are best addressed through the methods used to set harvest levels, biological reference points, etc.

11.4.2 How ecological linkages affect advice

Demonstrating that an ecological association exists does not, of itself, clarify how advice should take account of the linkage. The scale of its effect on advice can legitimately vary from minor to dominant, and some guidelines are needed for ensuring that it receives the proper weight in each case. Prior to discussing how to develop such guidelines, it is useful to consider the range of weightings given to ecological linkages in recent ICES advice.

11.4.2.1 Generally low effect on the advice – (MSVPA)

Multispecies virtual population analysis (MSVPA) has been implemented for the North Sea. It quantifies the predation mortality inflicted by each age of major fish predator (and, more recently, some seabirds and marine mammals) on each age of several prey, all of which are also exploited by fisheries. Some of these predation mortalities are high enough that they may meet future criteria for strong ecological linkages (ICES, 1996). Nonetheless, the Working Group recommended (ICES, 1987, 1988), and ACFM adopted, the practice that for single-species fisheries advice, it remains best practice to use the single-species assessment packages as a basis for annual harvest advice. The ecological interactions are captured adequately in the natural mortality term of the single-species assessment model. This parameter is based on MSVPA runs that are updated about decadally to capture large changes in abundances of the predators and prey in the North Sea.

The multispecies interactions that are captured by MSVPA are not always a minor part of scientific advice, however. The Multispecies Assessment Working Group (MAWG) stressed that if major changes in mesh sizes were to be considered, for example, their consequences should be evaluated using MSVPA (or another multispecies model), rather than single-species size-yield models (ICES, 1998). This is because the medium-term consequences of mesh size changes on the size composition of fish predators in the North Sea would redistribute and increase predation mortality in ways that changed greatly the estimated direct consequences of mesh size changes. Also, some predator-prey dependencies among marine fish are strong enough that ICES has concluded that they need to be captured directly in the assessment model used for single-species harvest advice, as is the case with Northeast Arctic cod (ICES, 2001a).

11.4.2.2 Generally one of several considerations in the advice – (herring and gravel beds)

The biological situation for herring has been described in Section 11.3.3.1. In an advisory context, advice on gravel beds is affected by many considerations and has been advanced from WGEXT and ACME. Where gravel beds are known to be used for herring spawning, this use is a dominant factor in advice on the management of gravel extraction. Where herring do not spawn, other factors might strongly influence advice on uses of gravel beds.

11.4.2.3 The dominant factor in the advice (black-legged kittiwakes and sandeels)

In the case of seabirds and sandeel fisheries (Section 11.2.3.1) the ecological dependency was the major determinant of the scientific advice on the specific management question. This reflects the specificity of the management question and the scientific advice as much as the strength of the ecological relationship. However, management of the sandeel fishery as a whole does consider the important role of sandeel as prey for many North Sea predators, and advice includes the ecological role both in the high predation mortality used in analytical assessments, and in the selection of biological reference points allowing for its role as prey, as well as just the SSB needed for recruitment. Similar considerations are also made in the reference points for Barents Sea capelin (ICES, 2001a). In these cases, factors other than ecological relationships do contribute to the scientific advice, but the ecological relationships are given a dominant role.

11.4.2.4 Developing a consistent advisory framework

From the preceding illustrations, it is clear that there is no single weight that ecological linkages should have in scientific advice and decision-making. Even simple rules such as very strong linkages should have great weight in the advice, whereas weak linkages should have only a little weight, might not provide good general guidance. Sometimes a request for advice might be quite specific to a situation where an overall weak ecological linkage was still an important factor in the advice, for example, advice on by-catch of a threatened or declining species, where by-catch events might be infrequent, but any mortality would be of concern. In other cases, even a very strong ecological linkage might be only one of several important considerations in advice, as with setting biological reference points for prey and for predators.

Notwithstanding the complexities, ICES would benefit from a clear set of guidelines (but not rigid rules) for assigning weight to ecological linkages when providing scientific advice. Such a set of guidelines would have the benefit of assuring that ACFM, ACME and ACE would all frame generally the same type of recommendations, given the same scientific information, when minor changes to the slant of a question might make any one of the groups most appropriate to deal with the the request in the first instance. This would reduce the need for MCAP to reconcile advice from different ICES Advisory Committees that might be perceived by clients to be inconsistent, because of the different emphases given to even particular phrasings by the different Advisory Committees. This, in turn, would aid the overall credibility of ICES advice to clients. WGECO is not able, in this meeting, to develop a full set of guidelines, and is unwilling to offer a partial and untested set, for fear that they might make practice worse rather than better. Nonetheless, we recommend that WGECO proceed with development of such a set of guidelines as a part of the process "to be able to consider ecological dependence in management advice."

11.4.3 Some practical considerations

11.4.3.1 Managing for species at risk – threatened and declining

A process is at present under way to select a set of threatened and declining marine species and habitats for protection as part of the implementation of Annex V to the OSPAR Convention. Criteria are being developed to select such species and habitats. An interim priority list is also under consideration (see Section 13, below). Regardless of precisely which species and habitats are selected under this process, it will be necessary for OSPAR Contracting Parties to take appropriate management steps to improve the situation for these species. ICES, in its role as scientific adviser to some of the most important management processes in the OSPAR area, should be in a good position to provide advice on the needs of many of the threatened and declining species. This advice might be from both the biological angle and from the management angle, especially for fisheries.

For threatened and declining species and habitats where fisheries contribute to the decline (OSPAR, 2001), WGECO could examine each case in detail, perhaps determining which particular fisheries are posing the greatest risk to the species/habitat. Possible mechanisms to reduce that risk might also be put forward with an assessment of likely efficacy (e.g. common skate would probably be best protected through the use of a large area of sea permanently closed to bottom fishing). ICES Working Groups most concerned with stocks utilised by the fisheries causing most impact could also contribute in their advisory suggestions. Finally, ACE and ACFM should combine their advice to provide information to fisheries managers and OSPAR.

WGECO could also provide a service by determining critical gaps in knowledge that might be filled through further research either on the species/habitat or on the fishery impact.

11.4.3.2 Managing vertical linkages: predator culls

Where there has been concern that predators are reducing prey to precarious abundances, or impeding recovery of their prey, directed programmes to reduce predator abundances ("culls") have been attempted. These have occurred primarily for terrestrial systems, although culls have been proposed for marine situations as well (see discussion in Punt and Butterworth, 1995; Punt, 1997). In a few cases, removal of predators has benefited the prey taxon as intended, but these cases have been highly controlled situations – commonly islands where an introduced exotic predator such as domestic cats or pigs have preyed on highly vulnerable resident species, such as breeding seabirds (e.g., van Rensberg, 1985; Jackson, 2001). In many more cases, however, the multiple linkages in food webs have meant that, although large reductions in predator populations have resulted in changes in prey populations as well, the medium- or long-term consequences of the predator reductions have not been as planned. In some cases, species other than the prey species intended to benefit from the predator removal have increased more and faster, resulting in greater competition rather than greater abundance for the intended beneficiary (Dahl-Hansen, 1995). In other cases, initial increases by the intended prey were unsustainable due to bottom-up effects, and when the food supply was depleted, the intended beneficiary population of the predator removal programme ended up with a smaller rather than larger population. In general, predator culls are an extremely high risk management option for benefiting specific prey populations, except in very special conditions.

11.4.3.3 Discards and offal

There have been many calls to reduce the amount of fish caught and discarded into the marine environment (e.g., 2002 Bergen Declaration). This is a laudable objective in relation to sustainable use of the marine environment and the general purpose of reducing wastage. However, a number of species are reliant on wastes from fishing vessels, especially scavenging species of birds (e.g., larger *Larus* gulls in the southern North Sea, Garthe *et al.* (1996)). Discarded fish form a reasonable part of the diet of some fish in some areas (Kaiser and Spencer, 1996). Changes in the amounts of fish discarded are likely to affect these scavenging species and this effect should perhaps be further evaluated. Such evaluations could consider various ways in which discards might be reduced and in which waste fish might be disposed of (e.g., comparisons of the effects of landing fish ashore with the effects of disposal at sea).

11.5 A workplan for ICES

Above, we have developed a framework for assessing "strong ecological linkages", but at this meeting WGECO had neither the time nor the expertise to implement this approach across the range of ecosystems within the ICES area. However, various ICES Working Groups contain the necessary expertise and so we propose a process (Table 11.5.1) in which this framework is applied by the relevant experts. Their opinions are fed back to WGECO where the material is integrated, combined with information on the scale of the threat from fishing activities and appropriate management measures and priorities considered. This consolidation and assessment is then reported to ACE for the formulation of advice to ICES customers.

As new requests are received or new information comes to light the process is initiated by ACE or WGECO, respectively (Step 4 in Table 11.5.1). In particular, this should occur when new scientific information comes to light or new societal views are expressed as to the level of change acceptable in the framework for a strong biotic interaction.

Table 11.5.1. The proposed process for the continued development of a mechanism for ICES to provide for "ecological dependence" in management advice.

Task	Lead party(ies)	Example timeframe
1. Development of a framework for assessing what constitutes a "strong ecological linkage"	WGECO	March 2002
2. Development of guidelines for Working and Study Groups to use the framework, to know when they are dealing with cases where "strong ecological linkages" are present	WGECO	2003
3. Development of guidelines for Working and Study Groups and Advisory Committees, to know what weight to give "strong ecological linkages" when drafting or finalising advice	WGECO and Advisory Committees and Working Groups	2003
4. New initiatives proposed, new scientific data emerge, new societal preferences expressed	ACE/WGECO	Spring Year 1
5. Specialists use framework for the gathering and assessment of information and commentary provided to WGECO on priority cases Biotic interactions Habitat interactions	 BEWG WGSE WGMMPH WGEF BEWG WGMHM	Autumn Year 1 – Spring year 2
6. Review of priority cases and analysis of ecological relevance, nature and strength of the linkages, and the threats to them	WGECO	May Year 2
7. Assessment of possible management responses for protection of ecological linkages and provision of commentary to ACE	WGECO	May Year 2
8. ACE formulates advice to ICES customers	ACE	Aug/Sept Year 2

11.6 References

Auster, P.J., Malatesta, R.J., and Larosa, S. 1995. Patterns of microhabitat utilization by mobile megafauna on the southern New England (USA) continental shelf and slope. Marine Ecology Progress Series, 127: 77–85.

Churchill, J.H. 1989. The Effect of Commercial Trawling On Sediment Resuspension and Transport Over the Middle Atlantic Bight Continental-Shelf. Continental Shelf Research, 9: 841–864.

Dahl-Hansen, G.A.P. 1995. Long-term changes in crustacean zooplankton—the effects of mass removal of Arctic charr *Salvelinius alpinus* from an oligotrophic lake. Journal of Plankton Research, 17: 1819–1833.

Emlen, J.M. 1983. Ecology: An Evolutionary Approach (2nd edition). Addison Wesley, Reading, MA.

Fosså, J.H., and Mortensen, P.B. 1998. Artsmangfoldet på *Lophelia*-korallrev og metoder for kartlegging og overvåkning (Diversity of species associated with *Lophelia* coral reefs and methods for mapping and monitoring) (in Norwegian with abstract and legends to tables and figures in English). Fisken og havet Nr. 17 – 1998. 95 pp.

Fosså, J.H., Mortensen, P.B., and Furevik, D.M. 2000. *Lophelia*-korallrev langs Norskekysten forekomst og tilstand. Fisken og Havet 2–2000. Havforskningsinstituttet, Bergen.

Fosså, J.H., Mortensen, P.B., and Furevik, D.M. in press. The deep-water coral *Lophelia pertusa* in Norwegian waters: distribution and fishery impacts. Hydrobiologia.

Frid, C.L.J., Harwood, K.G., Hall, S.J., and Hall, J.A. 2000. Long-term changes in the benthic communities on North Sea fishing grounds. ICES Journal of Marine Science, 57: 1303–1309.

Froese, R., and Pauly, D. (eds.) 2002. Fishbase. WWW publication, www.fishbase.org 23 March 2002.

Garthe, S., Camphuysen, C.J., and Furness, R.W. 1996. Amounts of discards in commercial fisheries and their significance as food for seabirds in the North Sea. Marine Ecology Progress Series, 136: 1–11.

Gislason. H. 1999. Single and multispecies reference points for Baltic fish stocks. ICES Journal of Marine Science, 56: 571–583.

Gray, J.S. 1981. The ecology of marine sediments: an introduction to the structure and function of benthic communities. Cambridge University Press, Cambridge. 185 pp.

Herrando-Perez, S. and Frid, C.L.J. 1998. The cessation of long term fly ash dumping: Effects on macrobenthos and sediments. Marine Pollution Bulletin, 36: 780–790.

ICES. 1996. Report of the Multispecies Assessment Working Group. ICES CM 1996/Assess:2.

ICES. 1998. Report of the Multispecies Assessment Working Group. ICES CM 1998/xxxx.

ICES. 2000a. Report of the ICES Advisory Committee on the Marine Environment, 1999. ICES Cooperative Research Report, 239: 7–10.

ICES. 2000b. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 2000/ACME:02. 93 pp.

ICES. 2001a Report of the ICES Advisory Committee on Fishery Management, 2001. ICES Cooperative Research Report, 246.

ICES. 2001b. Report of the ICES Advisory Committee on Ecosystems. ICES Cooperative Research Report No. 249. 94 pp.

Jackson, D.E. 2001. Experimental removal of introduced hedgehogs improves wader nest success in the Western Isles, Scotland. Journal of Applied Ecology, 38: 802–812.

Jensen, A., and Frederiksen, R. 1992. The fauna associated with the bank-forming deepwater coral *Lophelia pertusa* (Scleractinaria) on the Faroe shelf. Sarsia, 77: 53–69.

Kaiser, M.J., and Spencer, B.E. 1994. Fish scavenging behavior in recently trawled areas. Marine Ecology Progress Series 112: 41–49.

Kaiser, M.J., and Spencer, B.E. 1996. Behavioural responses of scavengers to beam trawl disturbance. *In* Aquatic predators and their prey. Ed. by S.P.R. Greenstreet and M.L. Tasker. Blackwell Science, Oxford.

Lindeboom, H., and de Groot, S.J. (Eds.). 1998. The effects of different types of fisheries on the North Sea and Irish Sea benthic ecosystems. European Commission, Brussels, IMPACT II Final Report. NIOZ-Rapport 1998-1. RIVO-DLO Report C003/98. 404 pp.

Maravelias, C. 1997. Trends in abundance and geographic distribution of North Sea herring in relation to environmental factors. Marine Ecology Progress Series, 159: 151–164.

Mayer, L.M., Schick, R.H., Findlay, R.H., and Rice, D.L. 1991. Effects of commercial dragging on sedimentary organic matter. Marine Environmental Research, 31: 249–261.

Nakken, O. 1994. Causes of trends and fluctuations in the Arctic-Norwegian cod stock. ICES Marine Science Symposium, 198: 212–228.

OSPAR. 2001. Summary record of the Workshop on Threatened and Declining Species and Habitats. BDC 01/4/2-E. OSPAR Commission, London.

Pearson, T.H., and Rosenberg, R. 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanography and Marine Biology Annual Review, 16: 229–311.

Pearson, T.H., and Rosenberg, R. 1986. Feast and Famine: Structuring factors in marine benthic communities. *In* Organization of communities: past and present, pp. 373–398. Ed. by J. H. R. Gee and P. S. Giller. Blackwell Scientific.

Punt, A.E. 1997. The effects of future consumption by seals on catches of Cape hake off South Africa. *In* Global trends in Fisheries Management. Ed. by E. Pikitch, D. Huppert, and M. Sissenwine. American Fisheries Society Symposium, 20. pp. 134–140.

Punt, A.E., and Butterworth, D.S. 1995. The effects of future consumption by the Cape fur seals on catches and catch rates of the Cape hakes. Modelling the biological interaction between Cape fur seals *Arctocephalus pusillus pusillus* and the Cape hakes *Merluccius capensis* and *M. paradoxus*. South African Journal of Marine Science, 16: 255–285.

Rhoads, D.C., and Boyer, L.F. 1983. The effects of marine benthos on physical properties of sediment: a successional perspective. *In* Animal-Sediment Relations, pp. 3–52. Ed. by P. L. McCall and M. J. S. Tevesz. Plenum, New York.

Rhoads, D.C., and Young, D.K. 1970. The influence of deposit-feeding organisms on sediment stability and community trophic structure. Journal of Marine Research, 28: 150–178.

Snelgrove, P.V.R., and Butman, C.A. 1994. Animal Sediment Relationships Revisited—Cause Versus Effect. Oceanography and Marine Biology, 32: 111–177.

Thrush, S.F., Hewitt, J.E., Cummings, V.J., Dayton, P.K., Cryer, M., Turner, S.J., Funnell, G.A., Budd, R.G., Milburn, C.J., and Wilkinson, M.R. 1998. Disturbance of the marine benthic habitat by commercial fishing: Impacts at the scale of the fishery. Ecological Applications, 8: 866–879.

Tuck, I.D., Hall, S.J., Robertson, M.R., Armstrong, E., and Basford, D.J. 1998. Effects of physical trawling disturbance in a previously unfished sheltered Scottish sea loch. Marine Ecology Progress Series, 162: 227–242.

*UKBAP. 2000. UK Biodiversity Group Tranche 2 Action Plans. Volume V – maritime species and habitats. English Nature, Northminster House PE1 1UA. ISBN 1 85716 467 9. 242 pp.

Vader, W., Barrett, R. T., Erikstag, K. E., and Strann, K.-B. 1990. Differential responses of common and thick-billed murres to a crash in the capelin stock in the southern Barents Sea. Studies in Avian Biology, 14: 175–180.

Van Rensberg, P.J.J. 1985. Feral cats and sub-Antarctic skuas on Marion Island: Competition or Coexistence. South African Journal of Science, 81: 11–19.

12 PROPOSED WORKSHOP ON (TOP-DOWN) ECOSYSTEM MODELS

Term of Reference: Review progress of activities initiated in 2001 by the Planning Group for a Workshop on [Top-down] Ecosystem Modelling.

The Planning Group for a Workshop on [Top-down] Ecosystem Modelling (PGEM) met from 6–8 March 2001 to plan comparisons of a range of ecosystem models. The PGEM was established at the request of WGECO (ICES, 1999), in response to the term of reference "to commence a review of the principal models of ecosystem dynamics and develop specific predictions based on them for the ecosystem effects of fishing".

The aims of the workshop proposed by PGEM were: 1) to make rigorous comparisons between different families of dynamic models that vary in their complexity or architecture and between model outputs and reality; 2) to examine the way models deal with and respond to uncertainty in their parameters and in data; and 3) to provide a "framework" that can be employed in the future for comparisons of other ecosystem models.

The outputs from the PGEM were reviewed by ACE in 2001. ACE concluded that it was desirable to complete comparison exercises between ecosystem models, and considered that such comparisons would yield information that would usefully inform an ecosystem approach to management and working groups such as WGECO. However, some minor reservations were expressed (ICES, 2001; Annex 4) and the following additions to the work of the PGEM were proposed:

1) to compile a comprehensive list of ecosystem modelling work in progress within ICES Member Countries;
2) to establish which comparisons between ecosystem models have already been completed;
3) to determine the requirements for additional comparisons between models;
4) subject to a need for additional comparisons between models, develop plans for a workshop at which models will be compared.

As a result, a request for further information on ecosystem model comparisons was circulated by the Chair of PGEM to all participants of PGEM. The response suggested that existing comparison exercises did not overlap with those proposed for the "Workshop on [Top-down] Ecosystem Modelling" and that the comparison exercise suggested by PGEM would provide useful new information.

WGECO continues to recognise the importance of comparing ecosystem models, and a workshop that allows the simultaneous comparison of models for the same areas is still required. To bring together appropriate model champions would require a funded workshop, and WGECO would support all efforts to find funding for this workshop.

References

ICES. 2000. Report of the Working Group on the Ecosystem Effects of Fishing Activities. ICES CM 2000 /ACME:02.

ICES. 2001. Minutes of the Advisory Committee on Ecosystems, 2001. ICES CM 2001/A:6.

13 THREATENED AND DECLINING HABITATS: ARE THE DATA SUFFICIENT?

A new term of reference given to WGECO shortly before the meeting, was as follows:

"Provide an assessment of the data on which the justification of the habitats in the OSPAR Priority List of Threatened and Endangered Species and habitats will be based; this assessment should be to ensure that the data used for producing the justification are sufficiently reliable and adequate to serve as a basis for conclusions that the habitats concerned can be identified, consistently with the Texel-Faial criteria, as requiring action in accordance with the OSPAR Strategy on the Protection and Conservation of the Ecosystems and Biological Diversity of the Maritime Area."

13.1 Introduction

It was made clear at the start of the meeting that WGECO should only be concerned with the habitats on the list, thus this section assesses the data upon which an initial list of habitats in need of protection measures, have been based. The purpose of this assessment is "to ensure that the data used are sufficiently reliable and adequate to serve as a basis for conclusions that the habitats concerned can be identified."

There has been recent activity in OSPAR to prepare a list of threatened and declining habitats, to contribute to the requirements of Annex V of the OSPAR Convention. In parallel with the on-going process to prepare and refine a set of robust selection criteria, the Contracting Parties to OSPAR were asked to submit a list of those habitats which they felt were already under threat or in decline, and which therefore needed immediate management action. Evaluation of these submissions and preparation of this list was dealt with intersessionally and considered by a workshop in Leiden in September 2001.

WGECO was asked to assess the data which were used to justify the inclusion of each habitat on the list. In the time available, and with limited time before the meeting to find relevant literature, the assessments of the status and threat of each habitat are not comprehensive. Where necessary, sections refer to the need for additional research or literature reviews to complete the evaluations, where they were seen to be insufficient.

It must be emphasised that the ToR asked us only to assess the data used to produce the list of habitats submitted to OSPAR, and not to provide comment on the suitability, or otherwise, of the criteria used to generate that list. We were also not asked to provide comment and suggestions for mitigation measures which may be necessary if these habitats are finally selected for management action. WGECO has a long history of advice on the appropriateness of mitigation measures for a range of human activities, and could apply these skills at a later date if required.

13.2 The OSPAR selection process

An initial list of threatened and declining habitats in the OSPAR maritime area was prepared by correspondence and reviewed at the Leiden workshop. Information was obtained using a questionnaire sent to each Contracting Party (see Table 13.2.1.1 for an example). Six Contracting Parties (Belgium, Iceland, the Netherlands, Norway, Portugal, Spain, and the UK) and two NGO Observers (BirdLife International and WWF) submitted replies. Although the selection process during the workshop did not include an extensive evaluation of each habitat using the Faial criteria (Table 9.2.1.a), these were tested by respondents during preparation of the questionnaire responses.

13.2.1 The outcome of the questionnaire submissions (Gubbay, 2001)

The results of the questionnaires were collated by Gubbay (2001), and this working paper was available to WGECO. Summary descriptions of each habitat type were prepared, and the quality of evidence presented on habitat status and threat was evaluated. Seventeen habitats were identified (Table 13.2.1.1).

Gubbay (2001) concluded that:

- The nominated habitats include examples that were found in 24 of the 26 biogeographic sub-divisions of the OSPAR Maritime Area.
- There was strong evidence presented on the status of 9 habitats.
- There was evidence relating to the decline of 9 of the nominated habitats.
- There was strong or reasonable evidence presented on the threats to 14 habitats.

Table 13.2.1.1. Evaluation of information submitted on habitats, with a description of quality of data for status and threat. This table has been copied directly from Gubbay (2001).

	DECLINE								THREAT		
	Potential characteristics for recording status	Reference level information available	Period of records/ assessment	Published information	Evidence of decline in OSPAR area	QUALITY OF EVIDENCE PRESENTED ON HABITAT STATUS		Potential threats identified	Reported effects	Effects on similar habitats	QUALITY OF EVIDENCE PRESENTED ON THREAT
Ampharete falcata sublittoral mud community	Extent, quality and species diversity/ abundance associated with the habitat	?	1986?	Yes	?	Unclear		Yes	?	Yes	Unclear
Carbonate mounds	Extent, quality and species diversity/ abundance associated with the habitat	Some areas?	1950s-2000	Yes	?	Unclear		Yes	Yes	Yes	REASONABLE
Deep sea sponge aggregations	Extent, quality and species diversity/ abundance associated with the habitat	Some areas?	?	Yes	?	Unclear		Yes	?	Yes	Unclear
Intertidal mussel beds	Extent, quality and species diversity/ abundance associated with the habitat	Detailed mapping of some sites	1980-2001	Yes	Yes	STRONG		Yes	Yes	Yes	STRONG
Intertidal mudflats	Extent, quality and species diversity/ abundance associated with the habitat	Detailed mapping of some sites	Since 1800s	Yes	Yes	STRONG		Yes	Yes	Yes	STRONG
Littoral chalk communities	Extent, quality and species diversity/ abundance associated with the habitat	Detailed mapping of some sites	Since 1980s	Yes	Yes	STRONG		Yes	Yes	Yes	STRONG
Lophelia pertusa reefs	Extent, quality and species diversity/ abundance associated with the habitat	Some areas	Very recent	Yes	Yes	STRONG		Yes	Yes	Yes	STRONG
Maerl beds	Extent, quality and species diversity/ abundance associated with the habitat	Detailed mapping of some sites	1970s	Yes	Yes	STRONG		Yes	Yes	Yes	STRONG
Modiolus modiolus beds	Extent, quality and species diversity/ abundance associated with the habitat	Basic mapping information	1950s-1990s	Yes	Yes	STRONG		Yes	Yes	Yes	STRONG
Oceanic ridges with hydrothermal effects	Extent, quality and species diversity/ abundance associated with the habitat	Some scientific study sites?	?	Yes	?	Unclear		Yes	Yes	?	STRONG
Ostrea edulis beds	Extent, quality and species diversity/ abundance associated with the habitat	Detailed mapping of some sites?	1900-2000	Yes	Yes	STRONG		Yes	Yes	Yes	STRONG
Sabellaria spinulosa reefs	Extent, quality and species diversity/ abundance associated with the habitat	Basic mapping information	1924-1982	Yes	Yes	STRONG		Yes	Yes	Yes	STRONG
Seamounts	Extent, quality and species diversity/ abundance associated with the habitat	Basic mapping information	?	Yes	?	Unclear		Yes	Yes	Yes	REASONABLE
Seapen & burrowing megafauna communities	Extent, quality and species diversity/ abundance associated with the habitat	?	Since 1980s	Yes	?	Unclear		Yes	?	Yes	Unclear
Soft bottom sediments on continental slope	Extent, quality and species diversity/ abundance associated with the habitat	?	1988; 1998	Yes	?	Unclear		Yes	Yes	Yes	REASONABLE
Sublittoral mud with seapens & burrowing megafauna of circalittoral muds	Extent, quality and species diversity/ abundance associated with the habitat	?	?	Yes	?	Unclear		Yes	Yes	Yes	REASONABLE
Zostera beds	Extent, quality and species diversity/ abundance associated with the habitat	Detailed mapping of some sites	1920-1930, 1987-2001	Yes	Yes	STRONG		Yes	Yes	Yes	STRONG

13.2.2 The outcome of the Leiden workshop: Four lists of habitats in need of protection

The working paper collating the results of the questionnaires (Gubbay, 2001) was reviewed by the Leiden Workshop. Using the available data the workshop prepared four lists:

<u>1) Threatened and/or declining species and habitats;</u>
Habitats with sufficient (i.e., strong or reasonable) evidence for a significant decline or threat.

<u>2) Species and habitats for which indications of serious decline and/or threats exist, but for which the status needs clarification;</u>
Habitats which showed clear indications of serious threat and/or decline, and for which the current status needed clarification.

<u>3) Priority threatened and/or declining species and habitats across their entire range within the OSPAR maritime area;</u>
Habitats which showed a severe decline or probability of significant decline, experienced a threat across most of its range within the OSPAR area, if its occurrence within the OSPAR area was of global importance, and if there were indications of serious threat in combination with a limited occurrence and/or a small recoverability.

4) Priority threatened and/or declining species and habitats for specific regions.
Habitats under severe decline or threatened within a specific OSPAR region(s).

It was recognised that while species and habitats on the priority lists (3 and 4) required urgent action, programmes and measures needed to be developed for all species and habitats on list 1.

Revision of the initial Gubbay (2001) list by the Leiden workshop resulted in a number of changes. An additional habitat, "Estuarine inter-tidal mudflats" was proposed. The three habitats "Sea pen & burrowing megafauna communities", "Soft-bottom sediments on continental slope" and "Sublittoral mud with sea pens & burrowing megafauna of circalittoral muds" were combined into a single habitat "Sublittoral mud with seapens & burrowing megafauna". The final list of 16 habitats resulting from the Leiden workshop and submitted to OSPAR BDC is shown in Table 13.2.1.2).

13.2.3 Data assessment

Each habitat type has been assessed by WGECO in order to determine the quantity and quality of supporting data. We have started by reviewing those on the priority list for the whole OSPAR area, and then reviewed the remaining habitats on the threatened and declining list (see Table 13.2.1.2). Each habitat description begins on a new page.

The first section "a) Description" provides a brief summary of the habitat type, threats that it may be experiencing and its current status. This text was taken directly from Gubbay (2001) and so is shown in italics.

Section "b) Literature used" lists the literature that was used as the basis for justification and cited in the questionnaire. These are identified in the reference list by *. This section also states whether WGECO felt that this was sufficient to support the classification.

Section "c) Literature interpretation" describes the WGECO opinion of whether the cited references have been interpreted fully and/or correctly.

Section "d) WGECO assessment" describes whether WGECO felt that the information and literature provided for location, decline and threats was sufficient, or whether there was other uncited literature that the group was aware of which could provide additional information. Additional uncited literature which contributes new data to the assesssment is cited here and listed in the final reference.

Finally, section "e) WGECO evaluation" provides an overall evaluation of the habitat made by WGECO and compares this conclusion with that for the Leiden workshop (OSPAR, 2001).

References

Gubbay, S. 2001. Review of proposals for an initial list of threatened and declining species in the OSPAR Maritime Area. Report to the National Institute for Coastal and Marine Management, the Netherlands.

OSPAR. 2001. Summary Record of the Workshop on Threatened and Declining Species and Habitats. BDC 01/4/2-E. OSPAR Commission, London.

Table 13.2.1.2. An initial list of threatened and/or declining habitats and the OSPAR-wide and region-specific priority list, based on OSPAR (2001). The section references to each habitat description are provided, and those in bold represent those which were considered to have highest priority. The column "Indication of decline …" is provided to show that this classification was available for selection, although in the Leiden assessment only species and no habitats were selected. In the column "Priority for specific regions", the Roman numerals refer to the OSPAR regions.

	THREATENED AND/OR DECLINING SPECIES AND HABITATS	INDICATION OF DECLINE AND/OR THREAT, FURTHER INFORMATION NEEDED	PRIORITY FOR WHOLE OSPAR AREA	PRIORITY FOR SPECIFIC REGIONS
Ampharete falcata sublittoral mud community	13.3.2.1			
Carbonate mounds	**13.3.1.1**		**13.3.1.1**	
Deep-sea sponge aggregations	**13.3.1.2**		**13.3.1.2**	
Intertidal mussel beds	13.3.2.2			
Marine intertidal mudflats	13.3.1.3			
Estuarine intertidal mudflats	**13.3.2.3**		**13.3.2.3**	
Littoral chalk communities	**13.3.1.4**		**13.3.1.4**	
***Lophelia pertusa* reefs**	**13.3.1.5**		**13.3.1.5**	
Maerl beds	13.3.2.4			
Modiolus modiolus beds	13.3.2.5			
Oceanic ridges with hydrothermal effects	**13.3.1.6**		**13.3.1.6**	
***Ostrea edulis* beds**	**13.3.2.6**			**II**
Sabellaria spinulosa reefs	13.3.2.7			
Seamounts	**13.3.1.7**		**13.3.1.7**	
Sublittoral mud with seapens and burrowing megafauna	**13.3.2.8**			**II, III**
***Zostera* beds**	**13.3.2.9**			**II, IV**

13.3 WGECO evaluation

13.3.1 Priority list of threatened and/or declining habitats in the OSPAR Maritime Area

13.3.1.1 Carbonate mounds

a) Description

Priority for whole OSPAR area.

"Carbonate mounds occur in small, localised clusters, mainly on the eastern margin of the North Atlantic. Most are dominated by filterfeeding communities and can support rich deep sea coral communities, which form secondary, biogenetic hard substrate for an abundant and diverse epibenthic fauna.

***Threats**: Although sound scientific information about carbonate mounds is scarce, it can be expected that benthic trawling operations have a serious mechanical impact from which the habitat and the associated ecosystem might not, or only very slowly, recover.*

***Status:** In areas where commercial benthic fisheries are carried out, there is a high probability of significant decline."*

b) Literature used (*below)

The literature used (WWF/IUCN 2001, OSPAR QSR for Region V) were not sufficient to support the nomination.

c) Literature interpretation

The references used provide no information on carbonate mounds as a biological habitat, only as geological features. There is no evidence in these references of either a threat to the geological feature itself, or to any decline. Mounds are referred to as structures on which *Lophelia* (see Section 13.3.1.5) and other corals grow. These reefs are threatened and declining (see Section 13.3.1.5), but there is no evidence that carbonate mound substrates are at any greater risk than other reef-supporting substrates (in fact, they may be at lower risk than, e.g., the sand mounds underlying the Darwin mounds).

d) WGECO assessment

This nomination requires additional literature if it is to be justified. However, no literature beyond that cited by WWF/IUCN (2001) was found. Research on carbonate mounds to the west of Ireland is under way in three EU-funded projects at present, but WGECO was unaware of any early results indicating threat to or decline of the mounds.

Location: The literature on the distribution of carbonate mounds indicates that these have been found only in OSPAR Region V off Ireland, but it is unclear whether mounds may exist elsewhere in the OSPAR area.

Decline: There is no literature on the decline in the extent of carbonate mounds.

Threats: There is no evidence of direct "clear and present" threats to the mounds. There is evidence of threat to biota growing on the mounds from fishing activities. It is conceivable that if these mounds formed by bacteria growing on hydrocarbon seepage (Peckmann *et al.*, 1998), then exploitation of that hydrocarbon may affect the structure.

e) WGECO overall evaluation

OSPAR (2001) considered this habitat to have priority for whole OSPAR area. The Leiden workshop concluded that evidence for the decline and threat were "unclear" and "reasonable", respectively. WGECO found no data on either a threat or a decline to the carbon mounds and concluded that there is insufficient evidence for the nomination.

References

*OSPAR Commission. 2000. Quality Status Report 2000, Region V – Wider Atlantic. OSPAR Commission, London 110 + xiii pp.

Peckmann, J., Reitner, J. and Neuweiler, F. 1998. Seepage related or not? Comparative analysis of phanerozoic deep-water carbonates. TTR7 Postcruise meeting Conference. Carbonate Mud Mounds and Cold Water Reefs: Deep Biosphere – Geosphere coupling. 27

*WWF/IUCN 2001. The status of natural resources on the high-seas. WWF/IUCN, Gland, Switzerland.

13.3.1.2 Deep-sea sponge aggregation

a) Description

Priority for whole OSPAR area.

"Deep sea sponge aggregations form a secondary, biogenetic hard substrate and are usually limited to small, restricted areas where hydrographic conditions are favourable. The habitat and its rich, diverse epibenthic fauna will recover only very slowly after being adversely affected.

***Threats:** Benthic fisheries and trawling operations can destroy sponge aggregations mechanically or by smothering them with sediments.*

***Status:** More information needed to assess the status of this habitat."*

b) Literature used (*below)

The literature used (OSPAR QSR for Region V) was not sufficient to support the nomination. The limited primary literature available was not used.

c) Literature interpretation

In the literature used, there were no references cited dealing specifically with the habitat in question.

d) WGECO assessment

The nomination requires additional literature if it is to be justified. More details describing the spatial extent of deep-sea sponge aggregations throughout the OSPAR area are required to justify their inclusion as threatened throughout the OSPAR region. Data are also required giving quantitative information on decline or threat.

Location: There are no reports available which give a comprehensive overview of the distribution of deep-sea sponge aggregations within the OSPAR area or from other waters. However, dense aggregations are known to occur in various places in the Northeast Atlantic (Klitgaard and Tendal, 2001). Deep-sea sponge aggregations are reported to occur close to the shelf break (250 m to 500 m depth) around the Faroe Islands (OSPAR Region I, Klitgaard and Tendal, 2001). Sponge aggregations have also been recorded along the Norwegian coast (OSPAR Region I) up to West Spitzbergen and Bjørnoya (Blacker, 1957; Dyer *et al.*, 1984; Fosså and Mortensen, 1998) and from the Porcupine Seabight (OSPAR Region III, Rice *et al.*, 1990)

Decline: No quantitative data on decline of sponge aggregations are available. Results of a questionnaire to local fishermen in the Faroe Islands indicate that, although such a habitat has existed in the past, fewer areas have now high concentrations of sponge aggregations (Klitgaard and Tendal, 2001).

Threat: There is no evidence of clear and present threats to deep-sea sponge aggregations. In terms of threat, the QSR mentions anecdodal reports indicating mechanical disturbance to biogenetic structures in general.

Vulnerability to future threat:

In the literature available no information is presented on future developments of threats. Information indicates that dominant sponge species are slow growing and take several decades to reach large size (Klitgaard and Tendal, 2001). In many areas, there is a common pattern of bottom trawling in increasingly deeper water where sponge aggregations are known to occur. Taking this into account, it seems reasonable to expect that the vulnerability and threat to the habitat is high.

e) WGECO overall evaluation

OSPAR (2001) considered this habitat to have priority for whole OSPAR area. The Leiden workshop concluded that evidence was "unclear", both for decline and threat. WGECO finds that there are no quantitative data on either a threat or decline to the habitat and concludes that there is insufficent evidence for the nomination. However, a single report from OSPAR region I indicates a decline.

References

Blacker, R.W. 1957. Benthic animals as indicators of hydrographic conditions and climatic change in Svalbard waters. Fishery Investigations (London), ser. II, 20: 1–49.

Dyer, M.F., Cranmer, G.J., Fry, P.D. and Fry, W.G. 1984. The distribution of benthic hydrographic indicator species in Svalbard waters, 1878–1981. Journal of the Marine Biological Association of the United Kingdom, 64: 667–677.

Fosså, J.H., and Mortensen, P.B. 1998. Artsmangfoldet på *Lophelia*-korallrev og metoder for overvåkning. Fisken og havet. No. 17, 95 pp.

Klitgaard, A.B. 1995. The fauna associated with outer shelf and upper slope sponges (Porifera, Demospongiae) at the Faroe Islands, northeastern Atlantic. Sarsia, 80: 1–22.

Klitgaard, A.B. and Tendal, O.S. 2001. "Ostur"-"cheese bottoms"-sponge dominated areas in Faroese shelf and slope areas. *In* Marine biological investigations and assemblages of benthic invertebrates from the Faroe Island, pp. 13–21. Ed. by G. Bruntse and O.S. Tendal. Kaldbak Marine Biological Laboratory, the Faroe Islands.

Klitgaard, A.B., Tendal, O.S., and Westerberg, H. 1997. Mass occurrences of large-sized sponges (Porifera) in Faroe Island (NE-Atlantic) shelf and slope areas: characteristics, distribution and possible causes. *In* The Responses of Marine Organisms to their Environments, pp. 129–142. Ed. by A.C. Jensen, M. Sheader, and J.A. Williams. Proceedings of the 30th European Marine Biology Symposium, University of Southampton, 362 pp.

*OSPAR Commission. 2000. Quality Status Report 2000, Region V – Wider Atlantic. OSPAR Commission, London 110 + xiii pp.

Rice, A.L., Thurston, M.H., and New, A.L. 1990. Dense aggregations of a hexactinellid sponge, *Pheronema carpenteri*, in the Porcupine Seabight (northeast Atlantic Ocean) and possible causes. Progress in Oceanography, 24: 179-206.

13.3.1.3 Marine intertidal mudflats

a) Description

Priority for whole OSPAR area.

"Mudflats are sedimentary intertidal habitats created by deposition in low energy coastal environments, particularly estuaries and other sheltered areas. Their sediment consists mostly of silts and clays with a high organic content. They are characterised by high biological productivity and abundance of organisms, but low diversity with few rare species. The largest continuous area of intertidal mudflats in the OSPAR area is in Region II bordering the North Sea coasts of Denmark, Germany and the Netherlands in the Wadden Sea and covering around 499,000 ha.

Threats: *Land claim for agricultural use has been a threat to this habitat in the past. Today it is more likely to be linked to maritime developments such as urban and transport infrastructure and for industry. Sea level rise is a current threat especially in areas where the land is sinking such as southern and south-east England, and any associated increased storm frequency, resulting from climate change, may further affect the sedimentation patterns of mudflats and estuaries. Fishing and bait digging can have an adverse impact on community structure and substratum, e.g., suction dredging for shellfish or juvenile flatfish by-catch from shrimp fisheries may have a significant effect on important predator populations.*

Status: *Reductions in the area of intertidal mudflats have occurred in many parts of the OSPAR area with locations at the heads of estuaries particularly favoured for land claim. A review carried out in the late 1980s noted that parts of at least 88 % of British estuaries had lost intertidal habitat to agricultural land claim in the past. Specific examples include the loss of over 80 % of the intertidal flat claimed for agriculture, industry and ports since 1720 in the Tees estuary, and in the Tyne estuary where no intertidal flats remain."*

b) Literature used (*below)

The literature that is cited as supporting this application (Doody *et al.,* 1991; Jones *et al.*, 2000; OSPAR Commission, 2000; UKBAP, 2000) is dominated by grey literature. The coverage of the primary literature by them is often very selective and restricted. The summary is also factually inaccurate, as there is one intertidal mudflat still present in the Tyne Estuary!

c) Literature interpretation

The majority of the literature is correctly interpreted, but in some cases the threats are exaggerated or used out of context.

d) WGECO assessment

There are sufficient primary sources accessible from the cited works to carry out the assessment.

Location: There is good evidence that intertidal mudflats occur throughout the OSPAR region and that the threats are similar in all areas.

Decline: The literature provides good evidence of declines in the extent of this habitat, but does not specify clearly whether the emphasis is on coastal intertidal mudflats or within estuarine habitats. This is necessary because the Leiden workshop also submitted a separate habitat "estuarine intertidal mudflats" (Section 13.3.2.3).

Threats: There are apparent threats to the existence of estuarine intertidal habitats. These arise from a range of activities from land claim, building of coast defences, sea level rise and coastal squeeze, pollution/waste disposal, fishing activities (particularly shellfish dredging and beam trawling), bait collection, and recreational visitors. A number of members of WGECO had extensive experience in the ecology of intertidal mudflats and the associated populations of birds and fish. The analysis presented here relies heavily on this knowledge of the original research studies as well as the regional context provided by the grey literature reports cited.

e) WGECO overall evaluation

OSPAR (2001) considered this habitat to have priority for the whole OSPAR area. The Leiden workshop concluded that evidence was "strong", both for the decline and threat. WGECO finds that there is good evidence of declines and threat to estuarine intertidal mudflats throughout the OSPAR region.

References

*Jones, L.A., Hiscock, K. and Connor, D.W. 2000. Marine habitat reviews. A summary of ecological requirements and sensitivity characteristics for the conservation and management of marine SACs. Joint Nature Conservation Committee, Peterborough. (UK Marine SACs Project report).

*Doody, J.P., Johnston, C. and Smith, B. (eds.) 1991. Directory of the North Sea Coastal Margine. Coastal Conservation Branch, Joint Nature Conservation Committee, Peterborough. 419 pp.

*OSPAR Commission. 2000. Quality Status Report 2000, Region II – Greater North Sea. OSPAR Commission, London 110 + xiii pp.

*UKBAP. 2000. UK Biodiversity Group Tranche 2 Action Plans. Volume V – maritime species and habitats. English Nature, Northminster House PE1 1UA. ISBN 1 85716 467 9. 242 pp.

13.3.1.4 Littoral chalk communities

a) Description

<u>Priority for whole OSPAR area.</u>

"The erosion of chalk exposures at the coast has resulted in the formation of vertical cliffs and gently sloping shore platforms with a range of micro-habitats of biological importance. Littoral fringe and supralittoral chalk cliffs and sea caves support algal communities unique to the substrate. The generally soft nature of the chalk results in the presence of

a characteristic flora and fauna, notably rock-boring invertebrates. Littoral chalk also supports distinct successive zones of algae and animals. Coastal exposures of chalk are a rare habitat in Europe with the greatest proportion (57 %) and many of the best examples of littoral chalk habitats located on the coast of England.

Threats: *The main threats to littoral chalk communities are from coast protection works, toxic contaminants and physical loss. Coast protection works have resulted in the loss of micro-habitats on the upper shore and the removal of splash-zone communities, including the unique algal communities. The deterioration of water quality by pollutants and nutrients has caused, respectively, the replacement of fucoid-dominated biotopes by mussel-dominated biotopes, and the occurrence of nuisance* Enteromorpha *spp. blooms. Sea level rise and post-glacial land adjustment will submerge areas of the intertidal chalk platforms.*

Status: *A recent survey of chalk cliffs throughout England revealed that 56 % of coastal chalk in Kent, and 33 % in Sussex have been modified by coastal defence and other works. On the Isle of Thanet (Kent) this increases to 74 %. There has been less alteration of chalk at lower shore levels except at some large port and harbour developments (e.g., Dover and Folkestone). Elsewhere in England, coastal chalk remains in a largely natural state."*

b) Literature used (*below)

The original references are referred to on the status of chalk habitats in OSPAR regions, and they contain specifc data which support the overall conclusion (Doody *et al.*, 1991; Laffoley *et al.,* 2000; UKBAP, 2000).

c) Literature interpretation

The conclusions cited in the literature are translated directly and accurately.

d) WGECO assessment

The literature quoted is convincing and it is not considered that further justification is necessary, although an assessment of the status of chalk communities elsewhere in European coastal waters would be helpful.

Primary literature on these habitats is very limited. However, the available primary and grey literature provide a good basis for assessing the extent and status of these habitats.

Location: Regional – the literature on the distribution of marine chalk habitats provides good coverage and clearly demonstrates that this environment is restricted to a limited number of locations in the OSPAR region.

Decline: There is limited literature on the decline in the extent of chalk habitats. It is clear from the available literature that some areas of habitat have been lost to development and coast protection works, but in many other areas the habitat has undergone a degree of modification.

Threats: There is a clear and present danger to the existence of these habitats. This comes primarily from physical threats such as development of ports or coast protection work and from water quality threats, including those arising from maritime accidents as many of the sites are in regions of high shipping activity.

The report considers these threats to be significant primarily as a result of the relatively restricted distribution and small total area of this habitat type. As a result, any loss must be regarded as significant in conservation terms. The available literature would confirm the factual basis of this statement but the "conservation significance" of any further loss is a matter of societal choice.

e) WGECO overall evaluation

OSPAR (2001) considered this habitat to have priority for the whole OSPAR area. The Leiden workshop concluded that evidence was "strong", both for the decline and threat. WGECO finds that there is good evidence of declines and threat throughout the OSPAR region.

References

*Doody, J.P., Johnston, C. and Smith, B. (eds.) 1991. Directory of the North Sea Coastal Margine. Coastal Conservation Branch, Joint Nature Conservation Committee, Peterborough. 419 pp.

*Laffoley, D. d'A., Connor, D.W., Tasker, M.L., and Bines, T. 2000. Nationally important seascapes, habitats and species. A recommended approach to their identification, conservation and protection. Prepared for the DETR Working Group on the Review of Marine Nature Conservation by English Nature and Joint Nature Conservation Committee. English Nature Research Report, No. 392, English Nature, Peterborough.

*UKBAP. 2000. UK Biodiversity Group Tranche 2 Action Plans. Volume V – maritime species and habitats. English Nature, Northminster House PE1 1UA. ISBN 1 85716 467 9. 242 pp.

13.3.1.5 *Lophelia pertusa* reefs

a) Description

Priority for whole OSPAR area.

*"*L. pertusa *has a wide geographical distribution, ranging from 55 °S to 70 °N and is present in the Atlantic, Pacific and Indian Ocean and in the Mediterranean, although most of the records come from the NE Atlantic. In Norwegian waters* L. pertusa *reefs occur on the shelf and shelf break off the western and northern parts on local elevations of the seafloor and on the edges of escarpments. The diversity of the taxa associated with the reefs is around three times as high as that of the surrounding soft sediment seabed, indicating that these reefs create biodiversity hotspots and increased densities of associated species.*

***Threats**: Offshore fisheries using bottom trawls are known to severely damage* L. pertusa *reefs. Corals are known to be susceptible to pollution and silting although the extent to which* Lophelia *may be affected is not clear at the present time. Petroleum industry developments with associated discharges of drilling mud and drill cuttings may negatively affect the corals.*

***Status**: The OSPAR area appears to be important for* L. pertusa *as a high proportion of the known occurrences of reefs are from this area. The widely scattered reported occurrences from other ocean areas indicate considerable uncertainty as to how well the distribution of* L. pertusa *has been mapped. Bottom trawling has destroyed large proportions of reefs along the Storegga shelf break and on banks on the Norwegian shelf. An assessment based on a study in Norway has indicated that approximately 30–50 % of coral reefs may have suffered some damage."*

b) Literature used (*below)

The following references were quoted by the questionnaire returns used by the Leiden workshop: UKBAP (2000); OSPAR BDC (2000); Dons (1944); Fosså and Mortensen (1998); Gubbay (2000?); OSPAR Commission (2000); Rogers (1999); Strømgren (1971); and WWF/IUCN (2001). These references are sufficient, especially OSPAR BDC (2000) that contains an annex written by two experts on *Lophelia* using many original references.

c) Literature interpretation

The conclusions cited in the literature are translated directly and accurately. There is additional evidence that *Lophelia* reefs off the UK (OSPAR Regions III and V) and the Faroes (OSPAR Region I) have also been destroyed by trawling, that is not quoted in the Leiden workshop summary. Reefs off Ireland (OSPAR Region III) are being affected by trawling also.

d) WGECO assessment

Primary literature on *Lophelia* reefs is extensive and growing rapidly. There is considerable further material available on this habitat (see, e.g., Section 7 of this report), but the material used in this assessment is sufficient and reliable without further support.

Locations: The distribution of *Lophelia* reefs in the OSPAR area is reasonably well known, although further surveys in some areas (e.g., Rockall and Hatton Banks, and the mid-Atlantic ridge) would improve this knowledge. The distribution covers all OSPAR Regions.

Decline: There is good evidence of decline in OSPAR Regions I, II, III and V. Occurrence in Region IV is not well known, but given the distribution of deep-water trawling it is likely that damage/decline has occurred there as well.

Threats: There is good evidence that the principal current threat comes from bottom trawling. As the technology to undertake such trawling in hard habitats develops further, areas of *Lophelia* reef have come under threat.

e) WGECO overall evaluation

OSPAR (2001) considered this habitat to have priority for the whole OSPAR area. The Leiden workshop concluded that evidence was "strong", both for the decline and threat. WGECO finds that there is good evidence of declines and threat throughout the OSPAR region.

References

*Dons, C. 1944. Norges korallrev. Det Kongelige Norske Videnskabers Selskabs Forhandlinger, 16: 37–82.

*Fosså, J.H., and Mortensen, P.B. 1998. Artsmangfoldet på *Lophelia*-korallrev og metoder for overvåkning. Fisken og havet No. 17, 95 pp.

*Gubbay, S. 2000? Offshore Directory. Review of a selection of habitats, communities and species of the North-East Atlantic. Report to WWF-UK.

*OSPAR BDC. 2000. Testing of the Faial criteria for the selection of species and habitats which need to be protected with the deep-water and habitat-forming coral reef species *Lophelia pertusa*. Paper 00/6/5 presented by Norway to OSPAR Biodiversity Committee in 2000.

*OSPAR Commission. 2000. Quality Status Report 2000, Region V – Wider Atlantic. OSPAR Commission, London 110 + xiii pp.

*Rogers, A.D. 1999. The biology of *Lophelia pertusa* (Linnaeus 1758) and other deep-water reef-forming corals and impacts from human activities. International Review of Hydrobiology, 84: 315–406.

*Strømgren, T. 1971. Vertical and horizontal distribution of *Lophelia pertusa* (Linne) in Trondheimsfjorden on the west coast of Norway. Det Kongelige Norske Videnskabers Selskabs Skrifter, 6: 1–9.

*UKBAP. 2000. UK Biodiversity Group Tranche 2 Action Plans. Volume V – maritime species and habitats. English Nature, Northminster House PE1 1UA. ISBN 1 85716 467 9. 242 pp.

*WWF/IUCN. 2001. The status of natural resources on the high-seas. WWF/IUCN, Gland, Switzerland.

13.3.1.6 Oceanic ridges with hydrothermal effects

a) Description

<u>Priority for whole OSPAR area.</u>

"*The hydrothermal vent fields of Menez Gwen, Lucky Strike, Rainbow and Saldanha are known important locations for these features. They cover very small areas in relatively shallow depths (compared to fields outside the OSPAR Maritime Area) of 850–2,300 m.*

Threats: *Scientific research and sampling activities can cause deliberate or accidental, long-lasting or irreversible damage to active chimneys and to the highly adapted, endemic fauna which depends on energy derived from sulphur-containing inorganic compounds.*

Status: *Most if not all known hydrothermal vent fields within the OSPAR maritime area are located in Region V. The ecological quality of these vent habitats might significantly decline if no protection or management measures are taken.*"

b) Literature used (*below)

The literature supports a comprehensive review of literature available to date published by WWF/IUCN (2001) and OSPAR Commission (2000).

c) Literature interpretation

The references used provide no information demonstrating the rate of decline or the extent of the habitat.

d) WGECO assessment

The reference quoted by OSPAR appears to have reviewed the bulk of the available literature.

Location: There is sufficient evidence to support the conclusion that such vents occur in OSPAR Region V, and indeed throughout the world.

Decline: Simply because our knowledge of the extent of these habitats is unknown, there is no empirical evidence to suggest that are in decline. As WWF/IUCN state in their 2001 report "Vast tracts of ridge crest remain unstudied and the abundance of vents is unknown; only 10 % of the 50,000 miles of ridge system has been explored."

Threat: This habitat has not been proven to be under threat from present-day human activities. WWF/IUCN (2001) cites no immediate potential for commercial exploitation (e.g. "to date, relatively few seafloor sulphide deposits have been shown to be of sufficient size and quality to be potential candidates for commercial exploitation") and suggests that any threats will occur as human technologies develop in the future. There is evidence presented for future threat to this habitat from sources of novel products for biotechnological applications, i.e., bioprospecting. (Jannasch, 1992), mining of polymetallic sulphide crusts within the next 10–15 years (Glowka, 1999), and tourism (Herring *et al.*, (1999) – although WGECO considers that this will be localised and of relatively low impact.

e) WGECO overall evaluation

There are insufficient data to support this nomination.

OSPAR (2001) considered this habitat to have priority for the whole OSPAR area. The Leiden workshop concluded that evidence for the decline and threat were "unclear" and "strong", respectively. WGECO finds that is no empirical evidence presented in the literature to suggest either decline or immediate threat to this habitat, although future threats could exist as human technology improves our capacity to reach them. WGECO concludes that there is insufficent evidence for the nomination.

References

*OSPAR Commission. 2000. Quality Status Report 2000, Region V – Wider Atlantic. OSPAR Commission, London 110 + xiii pp.

*WWF/IUCN. 2001. The status of natural resources on the high-seas. WWF/IUCN, Gland, Switzerland

13.3.1.7 Seamounts

a) Description

Priority for whole OSPAR area.

"Surrounded by abyssal plains, seamounts have special hydrographic/substrate conditions and act as 'islands' for epibenthic and pelagic faunas. They have a high rate of endemic species, are used as 'stepping stones' for the trans-oceanic dispersion of shelf species and as reproduction/feeding grounds for migratory species. Being of volcanic origin, the majority of seamounts lie along the Mid-Atlantic Ridge between Iceland and the Hayes fracture zone.

Threats: Seamount habitats are very sensitive to the physical impact of trawling and to the removal of benthic and pelagic key species by commercial fisheries. Being isolated and confined to small areas, seamount habitats and faunas will be able to recover only over long time periods by the sporadic re-colonisation from nearby seamounts and shelf areas. Where this is not possible (e.g., highly endemic species) disturbance might lead to extinction.

***Status**: Although there is a large seamount fishery in the Wider Atlantic, no information is available about the state of the seamount habitats/faunas in OSPAR Region V."*

b) Literature used (*below)

The nomination is not sufficiently supported, as the literature used (WWF/IUCN 2001, OSPAR QSR for Region V) does not cite primary references with original data regarding the OSPAR area.

c) Literature interpretation

The literature is incorrectly interpreted. The source used provides information in decline to seamounts biota in the Pacific but no information on either a threat to the biological habitats of seamounts within the OSPAR area or to their decline. Similarly, there is no evidence for threat to structural aspects of the habitat.

d) WGECO assessment

This nomination requires more supporting evidence if it is to be justified.

Location: The literature provided gives evidence that seamounts are found throughout the Atlantic Ocean, including OSPAR Region V.

Decline: There is no evidence of a decline in seamounts within the OSPAR area and information on decline of associated biota is related to areas outside the OSPAR area.

Threats: There is no evidence of direct threat to seamounts, and information on potential/actual threats to the biota they support is related to areas outside the OSPAR area.

e) WGECO overall evaluation

OSPAR (2001) considered this habitat to have priority for the whole OSPAR area. The Leiden workshop concluded that evidence for the decline and threat was "unclear" and "reasonable", respectively. WGECO finds no data on either a threat or decline to the seamounts within the OSPAR area, and concludes that there is insufficent evidence for the nomination.

References

*OSPAR Commission. 2000. Quality Status Report 2000, Region V – Wider Atlantic. OSPAR Commission, London 110 + xiii pp.

*WWF/IUCN. 2001. The status of natural resources on the high-seas. WWF/IUCN, Gland, Switzerland

13.3.2 List of threatened and/or declining species and habitats in the OSPAR Maritime Area

13.3.2.1 *Ampharete falcata* sublittoral mud community

a) Description

<u>Threatened and/or declining habitat for whole OSPAR area</u>

"*Habitat characterised by dense stands of* A. falcata *tubes which protrude from muddy sediments, appearing as a turf or meadow in localised areas. These areas seem to occur on a crucial point on a depositional gradient between areas of tide-swept mobile sands and quiescent stratifying muds. Dense populations of the small* Parvicardium ovale *occur in the superficial sediment. Substantial populations of mobile epifauna such as* Pandalus montagui *and small fish also occur, together with those that can cling to the tubes.*

Threats*: This biotope develops on undisturbed mud habitats. The main threats are therefore those that disturb the seabed such as benthic fisheries and seabed development.*

Status: *This biotope has been recorded on the seabed beneath the Irish Sea Front where* A. falcata *was found at densities of approximately 3,000/m^2 in 1986. Although there has been no detailed mapping showing change in extent, it is a biotope that is likely to have been affected by the activities described above. This has been reported in some personal observations".*

b) Literature used (*below)

The literature used was not sufficient to support the classification that the *Ampharete falcata* sublittoral mud community is threatened or declining across the whole OSPAR area. The only references cited are the QSR 2000 Region II and V reports, which do not refer to the habitat specifically and are not references to primary literature.

c) Literature interpretation

The literature was interpreted correctly insofar as evidence for both habitat status and presence of threat is unclear. It is not possible to draw the conclusion that the *Ampharete falcata* sublittoral mud community is threatened or declining across the whole OSPAR area from these reports.

d) WGECO assessment

While abundant information exists to demonstrate the detrimental effects of trawl fishing on a range of muddy benthic communities (de Groot and Lindeboom, 1994; Jennings and Kaiser, 1998; Jones *et al.*, 2000; Linnane *et al.*, 2000), none of the information cited in the working paper or other references available to WGECO illustrates the distribution of the habitat, or direct evidence of impact. While it is acknowledged that there is clear potential for these impacts to occur, there is insufficient evidence presented here to support this as a threatened and/or declining species.

Location: There is insufficient evidence to indicate the geographic distribution of this particular habitat within the OSPAR area.

Decline: There is insufficient evidence presented to indicate that this habitat is in decline.

Threats: There is insufficient evidence presented to argue that this habit is under threat, either immediately, or from future activities.

e) WGECO overall evaluation

OSPAR (2001) considered this habitat to be threatened and/or declining across the whole OSPAR area. The Leiden workshop concluded that evidence for the decline and threat of this habitat across the whole OSPAR area was "unclear". WGECO agrees that the evidence for both is insufficient.

References

de Groot, S.J., and Lindeboom, H.J. 1994. Environmental impact of bottom gears on benthic fauna in relation to natural resources management and protection of the North Sea. Reports of the Netherlands Institute for Sea Research, Texel.

Jennings, S., and Kaiser, M.J. 1998. The effects of fishing on marine ecosystems. Advances in Marine Biology, 34, 201–352.

Jones, L.A., Hiscock, K., and Connor, D.W. 2000. Marine habitat reviews. A summary of ecological requirements and sensitivity characteristics for the conservation and management of marine SACs. Joint Nature Conservation Committee, Peterborough. (UK Marine SACs Project report).

Linnane, A., Ball, B., Munday, B., van Marlen, B., Bergman, M., and Fonteyne, R. 2000. A review of potential techniques to reduce the environmental impact of demersal trawls. Irish Fisheries Investigations (New Series) No. 7. Marine Institute, Dublin. 39 pp.

*OSPAR Commission. 2000. Quality Status Report 2000, Region II – Greater North Sea. OSPAR Commission, London.

*OSPAR Commission. 2000. Quality Status Report 2000, Region V – Wider Atlantic. OSPAR Commission, London 110 + xiii pp.

13.3.2.2 Intertidal mussel beds

a) Description

Threatened and/or declining habitat across the whole OSPAR Region

"Intertidal mussel beds of M. edulis *are specific to the OSPAR region with the majority in the Wadden Sea and British coastal waters. Elsewhere different species form mussel beds. The beds are important in the sediment dynamics of coastal systems as well as being an important food source for birds. Mussel beds also provide shelter for a large number of organisms and form a rare hard substrate in a soft bottom environment.*

Threats*: The main threat to mussel beds is from fisheries. These are for seed mussels and occasionally on mature beds. When mature beds are destroyed recovery is difficult.*

Status*: Mature beds have been destroyed by fishing during periods with low spatfall and are only recovering very slowly, if at all. In the Wadden Sea, there has been no recovery of some areas in the last 12 years. Less than 10 % of the original area is now present."*

b) Literature used (*below)

The literature used is not sufficient to support the classification that intertidal mussel beds are declining or under threat across the whole OSPAR area, since most of the references used are from the Wadden Sea (Dankers *et al.*, 1999; Ruth. 1994; Ssymank and Dankers, 1996). The evidence for threat from fishing activities is sufficient with the literature used and is supported by other studies not cited here.

c) Literature interpretation

The literature used has been correctly interpreted regarding strong evidence of threat, but not for the entire OSPAR region, for which additional literature was required.

d) WGECO assessment

This assessment requires additional information on which to assess the status of this habitat. There is, however, a large amount of literature on this subject (see below) but it was not used or cited in the document. Further details are necessary to identify specific areas under threat from fisheries activity throughout the OSPAR region. Suggestions of possible references are:

Location: It is reported that mussel beds are present all along the coast of Europe (Jones *et al.*, 2000). Mussels occur in beds all around the UK (OSPAR Regions II and III) (Jones *et al.*, 2000), while in Germany, a series of surveys covering the whole littoral of Niedersachsen (OSPAR Region II, Germany) revealed a decrease in the extent of beds and more drastically in biomass (Dankers *et al.,* 1999). Dankers (1993) observed that beds in the Ameland region disappeared after intensive fisheries. Details on the mussel populations of Schleswig-Holstein for a period of nine years are also available (Ruth, 1994). Dankers *et al.* (1999) report a decrease in biomass of approximately 50 % between 1989 and 1990. The decline seems to be due to intensive fisheries and to low recruitment events. In France, mussel beds occur along the coast of France but no precise locations are cited (Jones *et al.*, 2000). In the Netherlands, Higler *et al*. (1998) observed a serious decline in the populations of mussels between 1988 and 1990, mainly caused by fisheries. The extent of mussel beds decreased from the 1970s to the 1990s and Dankers *et al.* (1999) suggest that the reduction was mainly due to intensive fisheries during a period of low recruitment events. In Denmark, intensive fisheries during 1984 to 1987 almost led to a complete disappearance of the mussel population (Kristensen, 1994, 1995).

Decline: Jones *et al.* (2000) and other literature cited here showed clear evidence for a decline of mussel beds in areas of intensive fisheries, especially when associated with low recruitment events. Intertidal mussel beds have been placed on biotopes and biotope complexes of the Wadden Sea (Ssymank and Dankers, 1996).

Threat: The extensive, heavily exploited mussel fisheries (especially spat collecting for aquaculture), such as in the Wadden Sea, removed close to the entire stock between 1988 and 1990 (Dankers *et al.*, 1999), resulting in increased mortality for seabirds (e.g., eider ducks) (Kaiser *et al.*, 1998) and affecting the benthic diversity. Jones *et al.* (2000), Dankers *et al.* (1999), and other reference consider that this habitat is under pressure from fisheries activities.

Vulnerability to future threat: Mussel beds are considered vulnerable to fisheries, especially when settlement of spatfall is low. It is well recognized that phytoplankton blooms, produced by nutrient enrichment (e.g., industrial and residential sewage discharge, agriculture), could have consequences on mussel beds. Nutrient enrichment and increase in phytoplankton production have been observed by de Jonge (1997) in the North Sea. Jones *et al.* (1999) suggested that mussel beds could also have intermediate sensitivity to anti-fouling substances and heavy metal contaminants. The decrease of mussel beds has profound effects on predators such as eider ducks and oystercatchers (Kaiser *et al.*, 1998).

e) WGECO overall evaluation

OSPAR (2001) considered this habitat to be threatened and/or in decline across the whole OSPAR area. The Leiden workshop concluded that evidence for the decline and threat of intertidal mussel beds was “strong” across the whole OSPAR area. WGECO has found sufficient evidence for the decline and threat of this habitat over the whole OSPAR area.

References

*Dankers, N. 1993. Integrated estuarine management—obtaining a sustainable yield of bivalve resources while maintaining environmental quality. *In* Bivalve filter feeders in estuarine and coastal ecosystem processes. Ed. by R.F. Dame. NATO ASI Ser., Subser. G: Ecological Sciences, Vol. 33: 479–511.

*Dankers, N., Herlyn, M., Sand Kristensen, P. Michaelis, H., Millat, G., Nehls, G., and Ruth, M. 1999. Blue mussels and blue mussel beds in the littoral. *In* Wadden Sea quality status report, pp. 141–145. Ed. by V.N. de Jong *et al.* Wadden Sea Ecosystem No. 9. Common Wadden Sea Secretariat.

de Jong, V.N. 1997. High remaining productivity in the Dutch western Wadden Sea despite decreasing nutrient inputs from riverine sources. Marine Pollution Bulletin, 34: 7427–7436.

Higler, B., Dankers, N., Smaal, A., and de Jong, V.N. 1998. Evaluatie van de ecologische effecten van het reguleren van schlpdievisserij in Waddenzee en Delta op bodemorganismen en vogels. *In* Structuurnota Zee- en Kustvisserij, van de maatregelen in de kustvisserij gedurende de erste fase (1993–1997). Appendix 5, pp. 17. Ed. by J.J. van Dijk and R. Heling.

Jones, L.A., Hiscock, K., and Connor, D.W. 2000. Marine habitat reviews: a summary of ecological requirements and sensitivity characteristics for the management of marine SACs. JNCC, Peterborough.

Kaiser, M.J., Laing, I., Utting, S.D., and Burnell, G.M. 1998. Environmental impacts of bivalve mariculture. J. Shellfish Res., 17: 59–66.

Kristensen, P.S. 1994. Blåmuslingebestanden i det danske Vadehav og Blåmuslingefiskeri (1991–1993). DFH-rapport nr. 476. 56 pp.

Kristensen, P.S. 1995. Blåmuslingebestanden i det danske Vadehav august 1995. DFH-rapport 1–96, 19 pp.

*Ruth, M. 1994. Untersuchungen zur Biologie und Fischerei von Miesmuscheln im Nationalpark-Schleswig-Holsteinisches Wattenmeer. Inst.f. Meeresforschung, Univ. Kiel, 327 pp.

*Ssymank, A. and Dankers, N. 1996. Red list of biotopes and biotope complexes of the Wadden Sea area. Helgoländer Meeresuntersuchungen 50, Suppl. 9–37.

13.3.2.3 Estuarine intertidal mudflats

This habitat was included in the outcome of the Leiden workshop, but had not previously been suggested or considered by Gubbay (2001) in her analysis of questionnaires submitted by Contracting Parties. There are clear overlaps between this habitat and ‘Marine intertidal mudflats’, which has been proposed for priority action and has been reviewed in Section 13.3.1.3. It has been suggested that threats to mudflats may be most acute within estuarine regions and, if this is so, the purpose of an additional habitat classification of ‘Marine intertidal mudflats’, is not clear. This habitat needs a precise definition before any further work is undertaken on a justificaiton for inclusion here.

13.3.2.4 Maerl beds

a) Description

<u>Threatened and/or declining habitat across the whole OSPAR area</u>

“*Maerl is a collective term for several species of calcified red seaweed. It grows as unattached nodules on the seabed and can form extensive beds in favourable conditions. Live maerl has been found at depths of 40 m but beds are typically much shallower, above 20 m and extending to the low tide level. They are an important habitat for a wide variety of marine animals and plants which live amongst or are attached to the branches, or which burrow in the coarse gravel or dead maerl beneath the top living layer.*

***Threats**: Commercial extraction of maerl is a threat to maerl beds in some areas, They are also threatened by benthic fisheries, fish farms and poor water quality. Studies have shown impact from scallop dredging for example, which caused serious declines of both maerl, by breaking and burying the thin layer of living maerl, and the associated species. Other types of mobile fishing gear are also likely to damage the living layer of maerl on top of the bed. In Brittany, eutrophication is known to have damaged maerl communities as this has caused smothering of the maerl by excess growth of other seaweeds and increased sedimentation. The discharge of nutrients from finfish farms into sealochs and the dispersal of chemicals used by fish farms into the marine environment may also affect the fauna associated with maerl beds.*

***Status**: Maerl is common on Atlantic coasts from Norway and Denmark in the north, to Portugal in the south. It is particularly abundant in Brittany. Spanish maerl deposits are confined mainly to the Ria de Vigo and Ria de Arosa. In Ireland, maerl is widely distributed in the south and south-west. It is absent from large areas of Europe, such as most of the North Sea, the Baltic, the Irish Sea and eastern English Channel, presumably due to environmental constraints. Major changes have been reported from some sites that have been studied in detail.”*

b) Literature used (*below)

The literature used (Birkett *et al.,* 1998; Jones *et al.*, 2000; Hall-Spencer and Moore, 2000; UKBAP, 2000) was not sufficient to support the classification that maerl beds are under threat or decline across the whole OSPAR region.

c) Literature interpretation

The literature used was not interpreted correctly. While strong indications of threat and decline in discrete areas were cited, this threat and decline were not indicated across the whole OSPAR area.

d) WGECO assessment

This classification is supported by hard evidence on the distribution of maerl beds across the entire OSPAR area but not by hard evidence of threat or decline on a large scale. It is therefore recommended that the classification be modified to be confined to OSPAR Region III until further hard evidence becomes available.

Location: The UK Biodiversity Action Plan reports that maerl beds occur off the southern and western coasts of the British Isles, north to Shetland. It also reports maerl beds occurring in other western European waters, from the Mediterranean to Scandinavia. This is supported by Jones *et al.* (2000), who report on maerl beds in the waters all around Europe. Birkett *et al.* (1998) report maerl over a broad geographic range, from Arctic Russia to the Mediterranean. The Leiden classification is therefore correct in referring to the entire OSPAR area.

Decline: Evidence that this habitat is undergoing decline at least in one small area is given in Hall-Spencer and Moore (2000), which recorded declines on a maerl bed off the west coast of Scotland related to the expansion of the scallop fishing industry there. Similar evidence exists off the Irish coast, where the situation was complicated as species came and went on maerl beds according to seasonal influences. It is therefore logical to suggest from the literature that maerl beds may be in decline as a result of various activities across the OSPAR region as a whole.

Threat: Evidence from the literature (Hall-Spencer and Moore, 2000) shows that a threat exists from scallop dredging activities on the beds studied. Evidence presented by Birkett *et al.* (1998) also describes threats from scallop dredging, as well as from extraction, suction dredging and pollution in nearshore waters. Evidence in the literature also states that maerl is slow growing in European waters and therefore slow to recover from disturbance, all of which supports the Leiden classification that this is a habitat under threat.

Vulnerability to future threat: If fishing activity with towed gears increases in future, then the threat to this habitat will increase. Furthermore, the threat also exists from extraction of maerl for pharmaceutical and other uses (De Grave *et al.*, 2000), from pollution by finfish and shellfish aquaculture operations in inshore waters, and suction dredging for bivalves.

e) WGECO overall evaluation

OSPAR (2001) considered this habitat to be threatened and/or declining over the whole OSPAR area. The Leiden workshop concluded that evidence for the decline and threat of maerl beds was 'strong' over the whole OSPAR area. WGECO agrees that evidence for decline and threat of this habitat is sufficient, but only for the OSPAR Region III area.

References

The following references provide useful additional information.

*Birkett, D.A., Maggs, C.A., and Dring, M.J. 1998. Maerl (volume V). An overview of dynamic and sensitivity characteristics for conservation management of marine SACs. Scottish Association for Marine Science (UK Marine SACs Project). 116 Pages.

De Grave, S., Fazakerley, S., Kelly, L., Guiry, M., Ryan, M., and Walshe, J. 2000. A study of selected maerl beds in Irish waters and their potential for sustainable extraction. Marine Resource Series, No. 10. Marine Institute, Dublin.

*Hall-Spencer, J.M., and Moore, P.G. 2000. Scallop dredging has profound, long-term impacts on maerl habitats. ICES Journal of Marine Science, 57: 1407–1415.

*Jones, L.A., Hiscock, K., and Connor, D.W. 2000. Marine habitat reviews. A summary of ecological requirements and sensitivity characteristics for the conservation and management of marine SACs. Joint Nature Conservation Committee, Peterborough. (UK Marine SACs Project report).

*UKBAP. 2000. UK Biodiversity Group Tranche 2 Action Plans. Volume V – maritime species and habitats. English Nature, Northminster House PE1 1UA. ISBN 1 85716 467 9. 242 pp.

13.3.2.5 *Modiolus modiolus* beds

a) Description

<u>Threatened and/or declining habitat across the whole OSPAR area</u>

*"The Horse Mussel (*M. modiolus*) forms dense beds at depths of 5–70 m in fully saline, often moderately tide-swept areas. Although it is a widespread and common species, true beds forming a distinctive biotope are much more limited. The composition of the biotopes is variable and is influenced by the depth, degree of water movement, substrate and densities, however, there can be an abundant epifauna and infauna and it has been considered to support one of the most diverse sublittoral communities in north-west Europe.*

Threats: *The main threat is from fishing, particularly using trawls and dredges. They are also likely to be badly damaged by other physical impacts such as aggregate extraction, trenching and pipe/cable-laying, dumping of spoil/cuttings or use of jack-up drilling rigs. The Horse Mussel is known to accumulate contaminants such as heavy metals in spoil disposal areas but the effects on condition, reproduction and mortality rates are unknown.*

Status: M. modiolus *is an Arctic-Boreal species whose distribution extends from the seas around Scandinavia and Iceland down to the Bay of Biscay. Within the OSPAR area it is particularly abundant in the Barents and White Seas, Iceland, Norway and the northern coasts of Britain. Scallop dredging is known to have caused widespread and long-lasting damage to beds in Strangford Lough, Northern Ireland and off the south-east of the Isle of Man."*

b) Literature used (*below)

The literature used was not totally relevant to supporting the case that this habitat is under threat or decline across the entire OSPAR region. The UKBAP (2000) is a description of the status of the *Modiolus* beds and did not discuss the sensitivity of this habitat. Furthermore, the OSPAR QSR for Region II never mentions specifically this species of mussel. However, Magorrian *et al.* (1995) observed damage to *Modiolus* beds in Strangford Lough, resulting from queen scallop trawling. These authors described three components in a bed. The most important component is the very rich community of sessile and free-living epifauna. The diversity of benthic species increased with the size and the number of mussel complexes (Ojeda and Dearborn, 1989). Jones *et al.* (2000) suggested from limited data that reef areas of mussel would have greater diversity of fauna than non-reef areas. Fishing activities, especially scallop dredging, have been found to damage a large amount of the epibenthic species living in association with *Modiolus* beds (Magorrian *et al.*, 1995; see Section 9.4.2 for more references on the impact of fishing activities on benthic epifauna). There is a need to identify the rate of recovery of this habitat after severe damage. There have been no studies on this subject.

c) Literature interpretation

The literature used was only partially interpreted correctly, arriving at the conclusion that evidence of habitat status is unclear. There was too much generality with the rare information available on the status of this habitat across the whole OSPAR area.

d) WGECO assessment

The classification as to "strong evidence presented on threat" in relation to the fragility of *Modiolus* beds in all OSPAR areas is not supported by the scientific literature available. However, some areas showed evidence of threat and the literature supports it. The need for more information on this habitat is essential and under the concept of precaution, the inclusion of this habitat should be considered as sensible, until more research on the status of this habitat is completed.

Location: Jones *et al.* (2000) reported that *Modiolus* beds occur in most areas of the Northern Atlantic, even if not listed in the Wadden Sea. Brown (1984) used some organisms in four different locations, north and south of Norway, Sweden and Ireland (OSPAR Regions I, II and III) for his study. *Modiolus modiolus* is a northern species and is more tolerant to low water temperature. For this reason, it seems that the southern limit of aggregation is around the British coast (Jones *et al.*, 2000). In the Wadden Sea, *Modiolus* is reported present but no detailed information is available (Jones *et al.*, 2000). In France, *Modiolus* beds occur along the coast of France but no precise locations are cited (Jones *et al.*, 2000). *Modiolus* is also present along the east coast of Canada and the USA. In the Gulf of Maine, the diversity of benthic organisms associated with *Modiolus* beds increased with the size of the bed (Ojeda and Dearborn, 1989)

Decline: From the literature used in the OSPAR report, there is no clear evidence of a decline in all areas mentioned above. The lack of information on the extent of this habitat and its actual status could be the cause of non-evidence of a decline for some areas. However, studies along the coast of the UK showed a clear decrease of this habitat (Magorrian *et al.*, 1995; Hill *et al.*, 1997). Jones *et al.* (2000) suggest that there is a significant decrease in the extent of this habitat. Furthermore, there is a shift from large long-lived benthic species to smaller and more opportunistic species (Lindeboom and de Groot, 1998).

Threat: Scallop dredging in the Strangford Lough considerably damaged *Modiolus* beds and the associated epifauna. Some areas could be under threat, but more information is needed.

Vulnerability to future threat: The biology of this species (long-lived and slow growing) places it in a vulnerable position, especially if you add the lack of information on its extent in the OSPAR area. Global warming and any phenomena (e.g., eutrophication) that increase the water temperature could also have an effect on this northern species.

e) WGECO overall evaluation

OSPAR (2001) considered this habitat to be threatened and/or declining across the whole OSPAR area. The Leiden workshop concluded that evidence for the decline and threat of *Modiolus modiolus* beds was "strong" across the whole OSPAR area. WGECO agrees that evidence for both decline and threat of this habitat is sufficient across the whole OSPAR area.

References

Brown, R.A. 1984. Geographical variations in the reproduction of the horse mussel, *Modiolus modiolus* (Mollusca, bivalvia). J. Mar. Biol. Ass. UK, 64: 751–770.

Hill, A.S., Brand, A., Veale, L.O.V., and Hawkins, S.J. 1997. The assessment of the effects of scallop dredging on the benthic communities. Contractor: Port Erin Marine Laboratory, University of Liverpool. MAFF Rep. No. CSA 2332. Feb. 97.

*Jones, L.A., Hiscock, K., and Connor, D.W. 2000. Marine habitat reviews. A summary of ecological requirements and sensitivity characteristics for the conservation and management of marine SACs. Joint Nature Conservation Committee, Peterborough. (UK Marine SACs Project report).

Lindeboom, H.J., and de Groot, S.J. 1998. IMPACT-II: The effects of different types of fisheries on the North Sea and Irish Sea benthic ecosystems. NIOZ rapport 1998-1. Den Burg, The Netherlands.

Magorrian, B.H., Service, M., and Clarke, W. 1995. An acoustic bottom classification survey of Strangford Lough, Northern Ireland. J. Mar. Biol. Ass. UK, 75: 987–992.

Ojeda, F.P., and Dearborn, J.H. 1989. Community structure of macroinvertebrates inhabiting the rocky subtidal zone in the Gulf of Maine: seasonal and bathymetric distribution. Mar. Ecol. Prog. Ser., 57: 147–161.

*OSPAR Commission. 2000. Quality Status Report 2000, Region II – Greater North Sea. OSPAR Commission, London.

UKBAP 2000 UK Biodiversity Group Tranch 2 Action Plans. Volume V - maritime species and habitats. Published by English Nature, Northminster House PE1 1UA. ISBN 1 85716 467 9. 242pp.

13.3.2.6 *Ostrea edulis* beds

a) Description

<u>Threatened and/or declining habitat</u>

"Oyster beds are found locally in estuarine areas from 0–6 m depth on sheltered but not muddy sediments, where clean and hard substrates are available for settlement. Populations also used to occur in deeper water, down to 50 m in the North Sea in places such as the Oyster Grounds. Juveniles usually settle on the shells of adult oysters so their decline reduces suitable settlement areas for subsequent generations.

***Threats:** Activities which disturb the seabed such as beam trawling and dredging have affected this habitat as well as overexploitation of the oysters themselves and the introduction of other (warm water) races of* O. edulis *and other oyster species in cultures.*

***Status:** Oyster beds were common in the North Sea and coastal waters. They have now virtually disappeared and only occur in some small remnant areas with populations occurring in deeper water in the North Sea, having disappeared gradually during the 19th and 20th centuries."*

b) Literature used (*below)

The reference used (UKBAP, 2000) adequately supports the nominations.

c) Literature interpretation

The literature is correctly interpreted.

d) WGECO assessment

Further details are necessary to identify the extent of *Ostrea edulis* throughout the OSPAR region.

Local: Native *Ostrea edulis* is widely distributed in Europe but seems to have disappeared from the Wadden Sea (OSPAR Region II) after overexploitation by the oyster fishery (Reise, 1982). *O. edulis* is cultured in the southwest of the Netherlands, in Norway (OSPAR Region I), along the coasts of Normandy and Brittany, Germany, and in several estuaries on the southeast coast of England (OSPAR Region II). *Ostrea* beds could be found in the rivers and flats bordering the Thames Estuary, The Solent, River Fal, the west coast of Scotland and Lough Foyle (OSPAR Region II).

The *Ostrea* population was thought to be extinct in the Wadden Sea after 1940 but a small number was found in 1992 (Dankers *et al.*, 1999). Furthermore, exotic species, such as *Crassostrea gigas*, are expanding in the German Wadden Sea and replacing the native *O. edulis* (Nehring, 1998).

Decline: A dramatic decrease in the population caused by fishery activities, again in the Wadden Sea, is bringing the population close to extinction. There are also reports of a reduction in the middle of the last century, which was caused by overexploitation. Other studies in North America have reached the same conclusion, which is that destructive harvesting and overfishing can reduce the habitat extent of oyster reefs. Most of the results showed a decline in the extent of natural oyster reefs.

Threat: The threat is present on the natural stock and a number of references indicate the cause as fisheries. There is a debate as to whether or not there is a truly natural UK stock. *O. edulis* has also been introduced in the Red List of Biotopes, Flora and Fauna of the Trilateral Wadden Sea Area.

Vulnerability to future threat: There is an increasing abundance of exotic species (*C. gigas*), which are more adaptable than *O edulis* in the German Wadden Sea. This shift in species could have profound effects on the natural stock of *O. edulis*. Furthermore, residential and industrial waste effluents could have consequences on mussel beds.

e) WGECO overall evaluation

OSPAR (2001) considered this habitat to be a threatened and/or declining habitat for OSPAR Region II. The Leiden workshop concluded that evidence was "strong", both for the decline and threat. WGECO finds that there is good evidence of declines and threat in OSPAR Region II.

References

Coen, L.D., Luckenbach, M.W., and Breitburg, D.L. 1998. The role of oyster reefs as essential fish habitat: A review of current knowledge and some new perspectives. Fish habitat: essential fish habitat and rehabilitation, 22: 438–454.

*Dankers, N., Herlyn, M., Sand Kristensen, P. Michaelis, H., Millat, G., Nehls, G., and Ruth, M. 1999. Blue mussels and blue mussel beds in the littoral. *In* Wadden Sea quality status report, pp. 141–145. Ed. by V.N. de Jong *et al.* Wadden Sea Ecosystem No. 9. Common Wadden Sea Secretariat.

Nehring, S. 1998. Natural processes in the Wadden Sea- challenging the concept of "an intact ecosystem". Ocean Challenge, 8: 27–29.

Reise, K. 1982. Long-term changes in the macrobenthic invertebrate fauna of the Wadden Sea: are polychaetes about to take over? Neth. J. Sea Res. 16: 29–36.

UKBAP 2000 UK Biodiversity Group Tranch 2 Action Plans. Volume V - maritime species and habitats. Published by English Nature, Northminster House PE1 1UA. ISBN 1 85716 467 9. 242pp.

13.3.2.7 *Sabellaria spinulosa* reefs

a) Description

Threatened and/or declining habitat across the whole OSPAR area

"Dense subtidal aggregations of this small, tube-building polychaete worm can form reefs at least several centimetres thick, raised above the surrounding seabed, and persisting for many years. They provide a biogenic habitat that allows many other associated species to become established and can act to stabilise cobble, pebble and gravel habitats. They are of particular nature conservation significance when they occur on sediment or mixed sediment areas where they enable a range of species that would not otherwise be found in the area to become established.

***Threats**: The greatest impact on this biogenic habitat is considered to be physical disturbance from fisheries activities. Dredging, trawling, net fishing and potting can all cause physical damage to erect reef communities. Aggregate dredging often takes place in areas of mixed sediment where S. spinulosa reefs may occur. Apart from direct removal, the impact of this activity on their long-term survival is unknown, but suspension of fine material during adjacent dredging activity is* *not considered likely to have detrimental effect. Pollution has been listed as one of the major threats to* S.spinulosa *in the Wadden Sea and may have partly contributed to their replacement by* Mytilus edulis *beds.*

***Status**: Research has attributed the loss of the large* S. spinulosa *reefs in the Wadden Sea to the long-term effects of fishing activity. A similar detrimental effect was reported for reefs in Morecambe Bay (UK) during the 1950s."*

b) Literature used (*below)

The literature used (Holt *et al.*, 1998; Hughes, 1998; Vorberg, 2000; UKBAP, 2000; and OSPAR QSR II, 2000) was not sufficient to support the classification that this habitat is under threat or in decline throughout the entire OSPAR area. The evidence for threat from fishing activities where it does occur is, however, sufficient.

c) Literature interpretation

The literature has not been fully interpreted correctly. The evidence on habitat status does apply to OSPAR Regions II and III, but not to the whole OSPAR area. Evidence presented on threat, however, does appear to be interpreted correctly.

d) WGECO assessment

This classification of the Leiden workshop for "strong evidence presented on habitat status" (i.e., geographic location and classification across the entire OSPAR area) does not appear to be supported by solid references, as only one reference reports that reefs of *S. spinulosa* are known from all European coasts, except the Baltic. A more robust classification might be to confine the classification to OSPAR Regions II and III. The Leiden classification as to "strong evidence presented on threat" as to the fragility of *S. spinulosa* in areas of trawl fishing, does however appear to be supported by the scientific literature cited.

Location: Information shows that *S. spinulosa* reefs are known from all European coasts, except the Baltic. In the UK, *S. spinulosa* colonies are reported to occur in discreet areas at a number of locations all round the English coast (OSPAR Regions II and III), mostly northwards of a line between the Bristol Channel and the Thames Estuary, as well as at one or two locations on the south coast, although not all of them are at sufficient density to be described as "reefs". On the German coast, intertidal reefs have been reported from the Wadden Sea (OSPAR Region II), where their absence is considered a good indicator of fishing intensity (Berghahn and Vorberg, 1993). The literature provided reports the occurrence of *S. spinulosa* on the French coast, but without precise locations.

Decline: The literature provided cites evidence for decline of *S. spinulosa* reefs in areas where trawl fishing occurs. *S. spinulosa* has been placed on the Red List of Macrofaunal Benthic Invertebrates of the Wadden Sea, according to the UK Biodiversity Action Plan, primarily due to beam trawling.

Threat: The literature provided cites a number of references indicating the threat to *S. spinulosa* reefs from fishing, and cites the practice by fishermen of destroying such reefs (as potential obstacles to trawls) with heavy gear prior to shrimp fishing, or to target such reefs as areas where shrimp might congregate. Also, the literature provided cites a number of references considering the risk from benthic trawling to be "high".

Vulnerability to future threat: Information indicates that *S. spinulosa* is very tolerant of water quality variation, but is potentially vulnerable to the short-term and localised effects of mineral extraction (although recovery from other, less-affected areas was predicted) and the effects of oil dispersants on the larvae. Overall, however, it has been concluded that *S. spinulosa* seemed unlikely to show any special sensitivity to chemical contaminants.

e) WGECO overall evaluation

OSPAR (2001) considered this habitat to be threatened and/or declining across the whole OSPAR area. The Leiden workshop concluded that evidence for both the decline and threat of *Sabellaria spinulosa* reefs was strong across the whole OSPAR area. WGECO agrees that evidence for both decline and threat to this habitat is sufficient, but only in OSPAR Regions II and III.

References

Berghahn, and Vorberg 1993. Effects of the shrimp fisheries in the Wadden Sea. Arbeiten des Deutschen Fischerei-Verbandes. Hamburg, 57: 103–126.

*Holt, T.J., Rees, E.I., Hawkins, S.J., and Reed, R. 1998. Biogenic reefs: An overview of dynamic and sensitivity characteristics for conservation management of marine SACs. Scottish Association of Marine Sciences (UK Marine SACs Project), Oban.

*Hughes, D.J. 1998. Sea pens, and burrowing megafauna (volume III). An overview of dynamic and sensitivity characteristics for conservation management of marine SACs. Scottish Association for Marine Science (UK Marine SACs Project). 105 pp.

*OSPAR Commission. 2000. Quality Status Report 2000, Region II – Greater North Sea. OSPAR Commission, London.

*UK BAP. 2000. Tranche 2 action plans. Volume V - maritime species and habitats. English Nature, Northminster House PE1 1UA. ISBN 1 85716 467 9. 242 pp.

*Vorberg, R. 2000. Effects of the shrimp fisheries on reefs of *Sabellaria spinulosa* (Polychaeta). ICES Journal of Marine Science, 57: 1416–1420.

13.3.2.8 Sublittoral mud with seapens and burrowing megafauna

a) Description

Threatened and/or declining habitat across the whole OSPAR area

"*The megafaunal burrowing activity creates a complex habitat, providing for surface enlargement and deep oxygenation. It is assumed that this type of habitat supports a much richer and/or higher biomass community of infauna.*

***Threats**: The deep muds are not easily affected but the extent of physical impact changes the habitat itself. Trawl tracks persist for prolonged periods in areas which are not tide or current swept. The main threats are therefore likely to be from demersal fisheries such as those which use beam trawls and scallop dredges.*

***Status**: The degree of decline is unknown but there is evidence that trawling does decrease and change the benthic communities living in this habitat."*

b) Literature used (*below)

The supporting literature (see below) contains sufficient information to support the classification that this habitat is declining/under threat in OSPAR Regions II and III.

c) Literature interpretation

The literature is correctly interpreted in terms of the geographic extent of the habitat (OSPAR Regions II and III). There is insufficient evidence on the extent of the threat to this habitat, however, but trawling activity in deeper waters is suspected.

d) WGECO assessment

WGECO would support the assessment regarding the status of the habitat and the "unclear" classification as to the quality of evidence presented on the threat, as well as the finding that "the degree of decline is unknown".

Decline: In spite of additional material researched by WGECO (Linnane *et al.*, 2000), evidence that this habitat is undergoing decline remains unclear, certainly for deeper water, simply because of gaps in our knowledge (although Roberts *et al.* (2000) reports evidence of deep-sea trawling physically impacting the seabed at depths of over 1000 m). Evidence from shallower waters (including Jennings and Kaiser, 1998) shows what damage communities of burrowing megafauna in muddy sediments endure as a result of trawling activities, that diversity of species is reduced, and how such communities can take several years to recover.

Threat: There is, however, robust evidence in the literature to support the classification that this habitat is under potential threat from trawling activities. Linnane *et al.* (2000) listed work giving estimates of penetration depth of up to 300 mm in mud for otter board trawl doors and beam trawls. Jennings and Kaiser (1998) also describe the detrimental effects of trawling on infauna in muddy habitats, as well as the effects of hydraulic dredges. They also point out that, in intensively fished zones (many of which occur in OSPAR Regions II and III), areas can be impacted several (over eight in the case of the southern North Sea) times a year. It is noted in the literature provided that fisheries for *Nephrops*, which themselves burrow in muddy habitats, can be intense and localised, which will increase the regularity of disturbance by trawl activity and therefore impact on the habitat itself.

Vulnerability to future threat: There is strong evidence in the literature to support the case that, as fishing effort increases, so will the threat to burrowing megafauna in sublittoral muds. As the activity of trawlers reaches further and further afield (Jennings and Kaiser, 1998), so will the threat to this habitat on a broader geographic scale than OSPAR Regions II and III, at which time it will be necessary to revisit the classification.

There is also evidence in the supporting literature to support the case that as human activity in the deep sea (such as deep-sea mining, hydrocarbon exploration) increases, so will the threat to deep-sea macrofauna from disturbance, which will be extremely slow (possibly several decades) to recover.

In addition to fishing activity, there is speculation in the literature provided that other threats, from heavy organic pollution and salmon aquaculture, may affect muddy habitats in shallow water.

e) WGECO overall evaluation

OSPAR (2001) considered this habitat to be threatened and/or under decline across the whole OSPAR area. The Leiden workshop concluded that evidence for the decline and threat was "unclear" and "unclear/reasonable", respectively. WGECO suggests that while the evidence of decline is insufficient, evidence for threat is sufficient across the whole OSPAR area.

References

*Auster, J.A., Malatesta, R.J., Langton, R.W., Watling, L., Valentine, P.C., Donaldson, C.L.S., Langton, E.W., Shepard, A.N., and Babb, I.G. 1996. The impacts of mobile fishing gear on seafloor habitats in the Gulf of Maine (Northwest Atlantic): implications for conservation of fish populations. Reviews in Fisheries Science, 4(2): 185-202.

*Bergmann, M., Beare, D.J. and Moore, P.G. 2001. Damage sustained by epibenthic invertebrates discarded in the *Nephrops* fishery of the Clyde Sea area, Scotland. Journal of Sea Research, 45: 105–118.

*Bluhm, H., Schriever, G., and Thiel, H. 1995. Megabenthic recolonisation in an experimentally disturbed abyssal manganese nodule area. Marine Geosources and Geotechnology, 13: 393–416.

*Borowski, C., and Thiel, H. 1998. Deep-sea macrofaunal impacts of a large-scale physical disturbance experiment in the Southeast Pacific. Deep Sea Research II, 45: 55–81.

*Hughes, D.J. 1998. Sea pens and burrowing megafauna (volume III). An overview of dynamic and sensitivity characteristics for conservation management of marine SACs. Scottish Association for Marine Science (UK Marine SACs Project). 105 pp.

*Jennings, S., and Kaiser, M.J. 1998. The effects of fishing on marine ecosystems. Advances in Marine Biology, 34: 201–352.

*Jones, L.A., Hiscock, K., and Connor, D.W. 2000. Marine habitat reviews. A summary of ecological requirements and sensitivity characteristics for the conservation and management of marine SACs. Joint Nature Conservation Committee, Peterborough. (UK Marine SACs Project report).

Linnane, A., Ball, B., Munday, B., van Marlen, B., Bergman, M., and Fonteyne, R. 2000. A review of potential techniques to reduce the environmental impact of demersal trawls. Irish Fisheries Investigations (New Series) No. 7. Marine Institute, Dublin. 39 pp.

*OSPAR Commission. 2000. Quality Status Report 2000, Region II – Greater North Sea. OSPAR Commission, London.

*OSPAR Commission. 2000. Quality Status Report 2000, Region III – Celtic Seas. OSPAR Commission, London.

*Roberts, J.M., Harvey, S.M., Lamont, P.A., Gage, J.D., and Humphrey, J.D. 2000. Seabed photography, environmental assessment and evidence for deep-water trawling on the continental margin west of the Hebrides. Hydrobiologia, 441: 173–183.

UKBAP 2000 UK Biodiversity Group Tranch 2 Action Plans. Volume V - maritime species and habitats. Published by English Nature, Northminster House PE1 1UA. ISBN 1 85716 467 9. 242pp.

13.3.2.9 *Zostera* beds (*Z. marina, Z. angustifolia* and *Z. noltii*)

a) Description

Threatened and/or declining in OSPAR Regions II and IV

"*Seagrass beds develop in intertidal and shallow subtidal areas on sands and muds. They may be found in marine inlets and bays but also in other areas, such as lagoons and channels, which are sheltered from significant wave action. They can survive and reproduce under conditions of occasional inundation or total submergence with the different species found at different shore levels or on different substrata. Seagrass stabilises the substratum as well as providing shelter and a substrate for many organisms. They are an important source of organic matter and may also be important nursery grounds, providing shelter for young fish.*

Threats*: Fisheries, nutrient inputs and turbidity (such as that which is caused by dredging activities) are all factors which threaten* Zostera *beds.*

Status*: All the* Zostera *areas are dramatically declining. In the Dutch Wadden Sea the area estimated to be covered by Zostera in the 19th century was believed to be between 90–150 km^2 (intertidal and subtidal). Today this has been reduced to less than 1 km^2 of intertidal bed*."

b) Literature used (*below)

The information on the status and decline of *Zostera* beds is sufficient for the UK and the Wadden Sea. However, the statement that "all the *Zostera* areas are dramatically declining" is not supported for the other areas within OSPAR Region II. The literature does not cover Region IV, but it does assess the east coast of the UK, which is part of Region II. The evidence for threat from nutrient inputs and turbidity is sufficient. The impact of fisheries is only documented for cockle fishing.

c) Literature interpretation

The literature is correctly interpreted, except for the spatial extent of *Zostera* decline.

d) WGECO assessment

More information is needed on the status and decline of *Zostera* in Region II and especially in Region IV.

Location: Davison and Hughes (1998), citing Stace (1997) and Cleator (1993), state that *Zostera marina* and *Z. noltii* occur throughout the whole Atlantic. *Z. angustifolia* is recorded only around the British Isles, Denmark and Sweden, which may be a matter of taxonomic disputes. According to Table 13.2.1.1, OSPAR focuses on Regions II and IV. However, there is no assessment of the present status that adequately covers this area.

The information on the status of *Zostera* beds is rather biased, with extensive literature on the UK (UKBAP, 2000; Davison and Hughes, 1998; Jones *et al.*, 2000; and www.marlin.ac.uk). The latter also describes a large bed of *Z. marina* in Irish waters, and Jones *et al.* (2000) also cover the Wadden Sea. There is no information available on other areas. Within the short time available, WGECO traced two sources of additional information. According to Geoffrey O'Sullivan (Marine Institute, Dublin, pers. comm.), Irish *Zostera* beds are stable, but no recent information is available for the last 20 years. Hendriksen *et al.* (2001) concluded that there are no clear trends in the development of *Z. marina* beds in Denmark (Region II) over the last 12 years.

Decline: The mass mortality of *Z. marina* owing to wasting disease during the 1920s and mid-1930s has been sufficiently described. More recently, declines have been reported in the Wadden Sea and the UK for both *Z. marina* and *Z. noltii* (Jones *et al.*, 2000; Davison and Hughes, 1998). No information is available on other areas.

Threat: Present threats, both natural and caused by human activities, are adequately compiled by Jones *et al.* (2000) and Davison and Hughes (1998). Actual impacts are recorded locally. The evidence for threat from nutrient inputs and turbidity is sufficiently described in both papers. Other well-documented threats are trampling (*Z. noltii*) and mechanical disturbance by boats (*Z. noltii* and *Z. marina*). The impact of fisheries is only documented for cockle fishing (Davison and Hughes (1998), citing Perkins (1988)).

Vulnerability to future threat: In the available literature, no information is presented on future developments of threats. However, given the long list of threats, the possibility of combined effects, and the long recovery time of affected beds, it seems reasonable to expect a great vulnerability of *Zostera* beds.

e) WGECO overall evaluation

OSPAR (2001) considered this habitat to be threatened and/or declining in OSPAR Regions II and IV. The Leiden workshop concluded that evidence for the decline and threat was "strong", respectively. WGECO finds that there is good evidence of declines and threat to this habitat. However, WGECO advises that the available literature only covers parts of Regions II and III; hence, a more robust classification might be to confine the classification to these Regions.

References

Cleator, B. 1993. The status of the genus *Zostera* in Scottish coastal waters. Scottish Natural Heritage, Review No. 22, Edinburgh.

*Davison, D.M., and Hughes, D.J. 1998. *Zostera* species: An overview of dynamic and sensitivity characteristics for conservation management of marine SACs. Scottish Association of Marine Sciences (UK Marine SACs Project), Oban.

Hendriksen, P., Andersen, J., Carstensen, J. *et al.* 2001. Marine områder 2000 – Miljøstilstand og udvikling. NOVA 2003. Danmarks Miljundersøgelser. 110 s. – Faglig rapport fra DMA nr. 375. http//faglige rapporter.dmu.dk (in Danish).

*Jones, L.A., Hiscock, K. and Connor, D.W. 2000. Marine habitat reviews. A summary of ecological requirements and sensitivity characteristics for the conservation and management of marine SACs. Joint Nature Conservation Committee, Peterborough. (UK Marine SACs Project report).

Perkins, E.J. 1988. The impact of suction dredging upon the population of cockles *Cerastoderma edule* in Auchencairn Bay. Report to the Nature Conservancy Council, South-west Region, Scotland. (No. NC 232 I).

Stace, C. 1997. New Flora of the British Isles. In prep.

UKBAP 2000 UK Biodiversity Group Tranch 2 Action Plans. Volume V - maritime species and habitats. English Nature, Northminster House PE1 1UA. ISBN 1 85716 467 9. 242pp.

13.4 Conclusions

13.4.1 Summary results of the WGECO assessment

WGECO was asked to assess the reliability and adequacy of the data on status, temporal trend and threat upon which the list of threatened and declining habitats was based. The outcome of this evaluation is presented in Table 13.4.1.1, together with the conclusions of Gubbay (2001) previously presented in Table 13.2.1.1.

In general, WGECO found that many of the justifications for selecting habitats were supported. It was apparent, however, that the literature cited in the original cases was often inadequate, referrring to generic references such as the Biodiversity Action Plans, or OSPAR QSR. Additional research by experts in the group found other literature which, in general, supported the cases made, and which, in fact, may have been familiar to those submitting the questionnaires. Table 13.4.1.1 shows that, having assessed the available information on threat and decline, WGECO found that it was generally sufficient to support the conclusions of the Leiden workshop, although with some revision. For example, WGECO felt that the case to support threat to the deep-sea sponge aggregations was insufficient for the entire OSPAR area, but sufficient for OSPAR Region I. Similarly, the supporting case for maerl bed habitats was thought to be sufficient for both threat and decline, but only for OSPAR Region III. While the case for *Sabellaria* was supported, it was considered that

WGECO could only support the case for OSPAR Regions II and III, rather than for the entire OSPAR area, as recommended by the Leiden workshop. Soft seabed habitats were combined into a generic group "Sublittoral mud with sea pens…..", and this extension of the scope has added complexity to the interpretation of threat and decline. In conclusion, it was felt that there was insufficient evidence of decline for any of these habitats, but that levels of potential threat were present and sufficient throughout the OSPAR area. The habitat had been proposed only for OSPAR Regions II and III. WGECO confirmed that there was sufficient evidence of potential threat to *Zostera* habitats throughout the OSPAR area, but that the available literature describing decline related only to Regions II and III, and suggested that the classification be restricted to these regions. The literature did not support evidence of decline in Region IV, as proposed at the Leiden workshop. WGECO disagreed with the outcome of the Leiden workshop and did not consider that there was sufficient evidence of threat to seamounts, or to oceanic ridges with hydrothermal effects. Estuarine mudflats were added late at the Leiden workshop and had not been specifically included in the questionnaires, although some respondents may have incorporated their views of threat and decline of estuarine mudflats into the category "intertidal mudflats", which was also available. WGECO felt that the justification for marine intertidal mudflats could be supported, but only in relation to estuarine habitats.

In several cases, the evidence for the spatial extent of the habitat was not available, and this has naturally led to a restriction in the OSPAR regions for which any mitigation measure may apply. Some habitats, especially those that occur in deep water, are almost certain to exist more widely but until research surveys confirm their distribution, further action cannot be taken. There is an implicit risk that further damage may occur by following this process.

WGECO emphasised that the summaries provided in Table 13.4.1.1 were only relevant to the evaluation of the published and grey literature that was made available to the group. There is still considerable effort required to clarify and revise the criteria that are used to select habitats that are under threat or in decline. The work necessary for this part of the process is achievable by WGECO, but requires a specific request. We interpreted our ToR to mean that we should avoid this wider debate relating to the criteria, and restrict our work to assessing the justfication used.

13.4.2 Comments on the OSPAR nomination process

To confirm that there are sufficient data to support the listing of each habitat, it is necessary to be aware of the criteria which were used in the process, and to which these data will be applied. For example, the Texel/Faial criteria (Table 9.2.1.a) identify quantitative levels for regional importance and decline of habitats, and this simplifies the process of evaluation of the supporting data for each habitat submission. It was noted that the category for significant decline (25–75 % decline in extent) was rather broad, and that declines in habitat extent of less than 25 % may also be significant.

It was beyond the terms of reference of the group to evaluate the appropriateness of the Texel/Faial criteria that were used in the selection process. It is necessary, however, to point out that the definition of the term "threat" was "clear and present" (OSPAR, 2001), which, while providing some guidance on how to use this criterion, still allowed some subjective interpretation. The guidelines attached to the Texel/Faial criteria make no specific reference to threat but allude to it under the sensitivity and decline criteria. This prevented the group from fully assessing whether supporting data describing "threat" justified the inclusion of a habitat.

Table 13.4.1.1. This table compares the assessment of the habitats presented by Gubbay (2001) as threatened and declining, with the result of the WGECO assessment of the data upon which the list was based, and described in detail in Section 13.3. The relevant section heading for each habitat is shown against each habitat type.

Habitat	Gubbay (2001)		WGECO	
	Decline	Threat	Decline	Threat
Ampharete falcata sublittoral mud community (13.3.2.1)	Unclear	Unclear	Insufficient	Insufficient
Carbonate mounds (13.3.1.1)	Unclear	Reasonable	Insufficient	Insufficient
Deep-sea sponge aggregations (13.3.1.2)	Unclear	Unclear	Insufficient [1]	Sufficient
Intertidal mussel beds (13.3.2.2)	Strong	Strong	Sufficient	Sufficient
Marine intertidal mudflats (13.3.1.3) Estuarine intertidal mudflats (13.3.2.3)	Strong	Strong	Sufficient [6]	Sufficient
Littoral chalk communities (13.3.1.4)	Strong	Strong	Sufficient	Sufficient
Lophelia pertusa reefs (13.3.1.5)	Strong	Strong	Sufficient	Sufficient

Maerl beds (13.3.2.4)	Strong	Strong	Sufficient [2]	Sufficient
Modiolus modiolus beds (13.3.2.5)	Strong	Strong	Sufficient	Sufficient
Oceanic ridge with hydrothermal effects (13.3.1.6)	Unclear	Strong	Insufficient	Insufficient
Ostrea edulis beds (13.3.2.6)	Strong	Strong	Sufficient	Sufficient
Sabellaria spinulosa reefs (13.3.2.7)	Strong	Strong	Sufficient [3]	Sufficient
Seamounts (13.3.1.7)	Unclear	Reasonable	Insufficient	Insufficient
Sublittoral mud with sea pens and burrowing megafauna of circalittoral muds (13.3.2.8)	Unclear	Unclear/ Reasonable	Insufficient [4]	Sufficient
Zostera beds (13.3.2.9)	Strong	Strong	Sufficient [5]	Sufficient

[1]WGECO found evidence of decline in OSPAR Region I.
[2]WGECO found evidence of decline in OSPAR Region III.
[3]WGECO found evidence of decline in OSPAR Regions II and III.
[4]Evidence of potential threat but little evidence of actual decline.
[5]Evidence of potential threat but little evidence of actual decline in Regions II and III
[6]Supported only for estuarine intertidal mudflats.

There is clearly a need to further refine the criteria that are used to select habitats for management action. However, this is also considered to be beyond the terms of reference of this section. The addition of criteria such as threat also requires further definition, in order to quantitatively assess when a threat become serious or significant, and evaluate how extensive a threat needs to be in relation to the spatial distribution of a habitat. Description of important features of "habitat" could usefully start with the Ecological Quality elements proposed in the Bergen Declaration for benthic organisms which include the density of opportunistic and sensitive (e.g., fragile) species (see Section 3.4.4). A further requirement would be for the inclusion of data on the physical integrity of the habitat, in terms of the extent of substrate available for key structuring organisms of the habitat. As part of this analysis, it is also necessary to understand the consequences of habitat fragmentation and the implications for habitat integrity.

13.4.3 Gaps in knowledge

It was apparent from this exercise that there is still much that needs to be done to develop our knowledge of marine habitats. This is especially evident for criteria such as Global and Regional "importance" (see Table 9.2.1.a for further details on this criterion), for which there are insufficient data. Also, the definition of habitat sensitivity is complex and it is unclear whether this should apply more to the structural and physical aspects of the habitat (chalk reefs which do not recover if damaged), or the individual species which occupy these substrates, and which themselves might recover relatively quickly.

It is inevitable that, with our current state of knowledge, there will still be issues which are unresolved and areas of research which do not provide complete answers. A precautionary approach to marine environmental management requires us to use best available information for assessing habitat status. It will therefore be necessary to apply knowledge gained from the results of impact on one habitat to another, on the assumption that similar responses may be seen. This is especially relevant to structural faunas which may undergo similar change when impacted by, for example, trawl or dredge gears.

Table 13.4.3.1. EXAMPLE Response - OSPAR questionnaire on threatened and declining habitat.

Species/Habitat: A5.131/B.-CorLop ***Lophelia pertusa* reefs**

Subspecies/Population:

OSPAR region: Entire OSPAR Area (N.E. Atlantic)

Biogeographic region: + Deep Sea - *Lophelia pertusa*-Reefs

National region:

Geographical extent of threat/decline: the Biogeographical region(s) indicated above

Seasonal Aspects: no

Significant seasons:

Significant areas in these seasons:

Importance Global: yes

Importance Global Specification:

Relative importance unknown due to lack of data, but so far data indicate that *L. pertusa* is of global importance:

The primary locations of *L. pertusa* are throughout the North Atlantic, including parts of West Africa, and persist down the sides of the Atlantic. It is also found in the Gulf of Mexico and the Caribbean and in some areas of the Pacific and Indian Oceans. *Lophelia* reefs have been found in the vicinity of cold-water seeps in the Gulf of Mexico and at the Lucky Strike hydrothermal vent field (10). It is also found in shallower waters (ca. 50 m deep) in the fjords of western Norway, off the Norwegian coast (e.g., Sula Ridge) and on the Swedish west coast (WWF/IUCN/WCPA 2001).

Importance Local/Regional: no

Importance Local/Regional Specification:

Rarity: no

Rarity Specification:

Sensitivity: Very sensitive

Sensitivity Specification:

from BDC 00/6/5:

L. pertusa is considered to be a sensitive species with respect to fishing activities with bottom trawls. It can also be considered to be very sensitive to fishing with heavy bottom gear, as extensive damage to coral reefs has been documented in Norway. The growth rate of the coral is very slow (5–15 mm per year) and the recovery time is therefore expected to be very long (centuries) for well-developed colonies. The sensitivity to other disturbances such as silting and pollution is not well known as there are few reported studies. Petroleum industry developments in deeper water may represent a threat for this species. *Lophelia pertusa* can be considered to be even more sensitive as a habitat than as a species. This is due to the time it takes for a large reef complex to develop, which could be several hundreds or even thousands of years.

Ecological Significance: no

Ecological Significance Specification:

from BDC 00/6/5:

L. pertusa may be considered a keystone species. It is habitat-forming, and the deep-water coral reefs harbour a rich diversity of other animal species. So far no species which occur obligatorily only on *Lophelia* reefs has been found. The species assemblages are therefore similar to those found also outside the reefs. Many species, however, occur in much higher abundance on coral reefs than on other habitats. The deep-water *Lophelia* reefs are important habitats for some fish species. Observations of gravid females of *Sebastes marinus* indicate that the coral reefs may play a role in the reproduction of this species.

Keystone Species: no

Keystone Species Specification:

Decline:

Decline Specification:

from BDC 00/6/5:

An assessment based on a study in Norway has indicated that approximately 30–50 % of coral reefs may have been more or less damaged mainly because of bottom trawling. This may have been the cumulative impact of trawling over many decades. It is likely that the situation is similar in other parts of the OSPAR area with similar bottom trawl fisheries. It is therefore likely that *L. pertusa* has had a significant decline in the OSPAR area.

Species Requirements:

Threat:

- Demersal fishing:

Physical impact by bottom trawling is the main threat to date: Much of the coral habitat in the N.E. Atlantic region coincides with suitable seabed for trawling operations. Deep-sea trawlers operate to depths of 1,900 m. Reef damage has been documented in many areas and is likely to have occurred in many more areas. Evidence of the extent of impacts of trawling is rather limited although side-scan sonar surveys from the eastern Porcupine Seabight and from the continental slope West of Shetland show evidence of trawl scouring on the seabed and damage to coral (17,18). As *L. pertusa* occurs in these same regions, it is highly likely that it has been impacted heavily by deep-sea fishing. The total destruction of some *L. pertusa* reefs as a result of demersal trawling has been reported in shallower Norwegian waters around Storegga and present estimates suggest that 30–50 % of Norwegian reefs have already been damaged or destroyed(15). Growth rates of *L. pertusa* have been estimated from 4–25 mm per year (19) similar to some massive shallow-water coral species but slower than other massive branching corals. Slow growth may limit or prevent its recovery from reef damage. (WWF/IUCN/WCPA 2001 and references therein)

- Pollution:

Oil exploration and production is currently occurring near the areas where *L. pertusa* reefs are abundant in the N.E. Atlantic. Discharges of drilling mud and drill cuttings from these activities may negatively affect the corals. It is difficult to predict the area of drill cutting and mud dispersal and hence the magnitude of the impact upon local coral communities (3). Studies in shallower waters have shown that contaminants from oil platforms may be detected in significant quantities up to 6,000 m from the installation, covering an area up to 100 km^2 (20). In deeper water operations where there is surface discharge of contaminants, the physical extent of contamination may be far greater (3). In shallow water corals, drill cutting exposure has been shown to cause coral death, alter feeding behaviour, alter coral physiology and induce morphological changes (3). In addition to drill cuttings, oil contamination has been shown to affect shallow-water coral communities, having toxic effects on corals resulting in reduced growth, tissue damage, disruption of cell structure, damage to stimuli response and feeding behaviour, excessive mucus production, alterations to reproductive success or mortality (21).

Recovery of *L. pertusa* reefs and their communities from the potential impacts of oil exploration and production may be extremely slow (3) (WWF/IUCN/WCPA 2001 and references therein).

Data Sources:

BDC 00/6/5: Testing of the Faial criteria for the selection of species and habitats which need to be protected with the deep-water and habitat-forming coral reef species *Lophelia pertusa*. Presented by Norway.

Gubbay, S. (in press). Offshore Directory. Review of a selection of habitats, communities and species of the North-East Atlantic. Report to WWF UK.

Rogers, A. 1999. The biology of *Lophelia pertusa* and other deep-water reef-forming corals and impacts from human activities. Int. Review of Hydrobiology, 84(4): 315–406

WWF/IUCN/WCPA (2001).The status of natural resources on the high seas.
WWF/IUCN Gland, Switzerland

Assessment Period: all facts are very recent, and ever more destroyed reefs are discovered – no time series necessary.

Contact person: Sabine Christiansen

Contact address:

WWF International

Northeast Atlantic Programme

Am Guethpol 11

28757 Bremen

Germany

Contacts phone number: +49(0)421-658 4628/+49407424697

Contacts fax number: +49(0)421-658 4612/+49407424697

Contacts E-mail: christiansen@wwfneap.org

Institute website: http:/www.wwfneap.org

Other contacts:

Notes:

Submitted/updated on: 13-7-2001

Contracting Party / Observer: NGO

References

Gubbay, S. 2001. Review of proposals for an initial list of threatened and declining species in the OSPAR Maritime Area. Volume 1 - review.

OSPAR 2001. Summary Record of the Workshop on Threatened and Declining Species and Habitats. OSPAR Commission, London, BDC 01/4/2-E.

14 FUTURE ACTIVITIES AND RECOMMENDATIONS

14.1 Recommendation: Terms of Reference

Recommendation: Terms of Reference for WGECO should be stated clearly and unambiguously.

Justification:

Members of WGECO have found it increasingly difficult to interpret the WGECO Terms of Reference. If WGECO received clear and unambiguous terms of reference, it would have saved the group both time and effort. It would be helpful if an appropriate member of the Secretariat were available at the start of the meeting to introduce the Terms of Reference and to answer related queries. Moreover, it would be helpful if Terms of Reference that were added late in the process at the Statutory Meeting (or after the Statutory Meeting) were accompanied by an explanation and introduced in detail to the group.

14.2 Recommendation: Publication of an ICES CRR

WGECO recommends that the former Chair, Jake Rice, edits a summary document integrating the results presented in the last three working group reports (1998–2001) by topic and that this document is published in the *ICES Cooperative Research Report* series.

Justification:

Over the past years, the work of the Working Group on Ecosystem Effects of Fishing Activities has been an amalgamation of responses to requests for advice and products of an internal drive to make scientific progress in this field. Progress on both types of work often covers a time span of more than one meeting, but the coherence is often somewhat lost because results are distributed over several reports. WGECO feels that it would be extremely useful if the important results obtained under the chairmanship of Jake Rice could be made available in an integrated and properly edited form to the broader ICES and non-ICES audience. The *ICES Cooperative Research Report* series would be the obvious choice.

The CRR will trace the development of several key issues over the span of these three meetings. Topics will include indicators of ecosystem status, concepts and practices in setting reference points for ecosystem properties other than target species of fisheries, and possibly others, depending on examination of what information is available in Advisory Committee reports already. In addition, where WGECO addressed a topic of high profile, and provided clear scientific conclusions about the scale of the issue and proposed appropriate management actions, that information will also be presented. Topics here include impacts of mobile fishing gears on benthic ecosystems, the way forward with Ecological Quality Objectives, and a number of related issues.

14.3 Recommendation: Provision of access to satellite vessel monitoring data

WGECO recommends that ICES endeavour to obtain and collate national satellite vessel monitoring data for the ICES area/ OSPAR regions.

Justification:

Detailed data on the spatial and temporal distribution of fishing effort are required to identify habitats that are potentially impacted by fishing and to determine the frequency of those impacts. International fishing effort data on appropriate scales are best provided by the satellite vessel monitoring systems that have been introduced by national governments in recent years, and a compilation of national satellite vessel monitoring data would have allowed WGECO to provide significantly more comprehensive responses to questions on the advice needed for an EcoQO framework (ToR “a”), comparative impacts of human activities on ecosystem dynamics and nutrient turnover (ToR “b”) and the sensitivity of species and habitats to bottom fishing impacts (ToR “c”, “e” and “g”).

14.4 Recommendation: Habitat mapping

Recommendation: WGECO recommends that ICES should attempt to facilitate the production of comprehensive small-scale habitat maps for the ICES area / OSPAR regions.

Justification:

Detailed data on the spatial distribution of marine habitats are essential for identifying the scale and significance of human impacts in the marine environment. Access to comprehensive but small-scale habitat maps for the ICES/ OSPAR region, in conjunction with detailed data on the spatial and temporal distribution of human impacts (see Recommendation 3), would have enabled WGECO to make significantly greater progress in quantifying the comparative impacts of human activities on ecosystem dynamics and nutrient turnover (ToR “b”) and assessing the scale of bottom fishing on species and habitats (ToR “c”, “e” and “g”).

14.5 Recommendation: Continued exploration of ecosystem metrics

Recommendation: that WGECO be given a term of reference to continue the exploration of the effect of fishing activities on fish assemblages and marine ecosystems with particular focus on (i) the exploration of spatial analysis methods for assessing ecosystem properties. (ii) the further investigation of the suitability of the metrics examined in 2002 for use in the support of scientific advice in the context of an ecosystem approach to management.

Justification:

(i) Previous work on metrics to assess the impacts of fishing on fish communities has generally focused on summary statistics derived from annual surveys. Recently, attempts have been made to utilize the spatial pattern of fisheries effort and disaggregated survey information to test hypotheses regarding fishery impacts. While this approach has considerable potential, it is clear that there are unresolved methodological issues. Disaggregated analyses of survey data also have potential to yield metrics of other properties of the ecosystems such as spatial integrity.

(ii) Work this year has shown the potential of metrics based on size spectra to offer a robust response to fishing-induced changes. Other metrics appeared to show responses that were either sensitive to the nature of the data sets used or to differences in the inherent structure and dynamics of the various ecosystems. There is clearly a need for further work to understand these metric’s behaviours.

14.6 Recommendation: Analytical workshops on the ecosystem effects of fishing activities

Recommendation: that consideration be given to WGECO adopting a model similar to that used by ACFM, such that Terms of Reference (such as Term of Reference (c) in 2002 and any following arising from Recommendation 14.5) that require substantial analysis be tasked to a workshop to be held immediately preceding the meeting of WGECO.

Justification:

In recent years, our workload has increased dramatically and the proportion of work which is urgent by dint of needing to help ICES formulate advice has also grown. It is increasingly difficult for WGECO to give due attention to analytical and exploratory Terms of Reference while giving the necessary time and effort to the advisory Terms of Reference. By moving the more exploratory analyses to a workshop prior to the meeting, time is freed up for the working group. Running the workshop immediately prior to WGECO will ensure that results are available and utilized by WGECO in the formulation of scientific work to underpin advice and members can keep travel costs to a minimum. This model has been used by ACFM in the past with some success.

ANNEX 1: LIST OF PARTICIPANTS

18–27 March 2002

Name	Institute/Address	Telephone No.	Fax No.	E-mail
Philippe Archambault	Institut Maurice-Lamon-tagne Fisheries and Oceans Canada 850 Route de la Mer, Mont Joli (QC) G5L 4M5 Canada	+1 418 775-0718	+1 418 775-0752	archambaultp@dfo-mpo.gc.ca
Niels Daan	Netherlands Institute for Fisheries Research Haringkade 1 P.O. Box 68 NL-1970 AB IJmuiden Netherlands	+31 255 564 693	+31 255 564 694	niels@rivo.wag-ur.nl
Andrey Dolgov	Polar Research Institute of Marine Fisheries and Oceanography (PINRO) Knipovich Str. 6 Murmansk 183763 Russia	+7 8152 47 24 69 and +7 8152 47 24 62		dolgov@pinro.murmansk.ru
Martin Dorn	Alaska Fisheries Science Center, Seattle, Washington 98115-0070, USA	+206 526 6723	+206 526 6548	martin.dorn@noad.gov
Lisette Enserink	Nat. Institute for Coastal and Marine Management P.O. Box 20907 2500 EX Den Haag Netherlands	+31 70 311 4363	+31 70 311 4380	e.l. enserink@ rikz.rws.minvenw.nl.
Chris Frid (Chair)	University of Newcastle Dove Marine Laboratory Cullercoats North Shields NE30 4PZ United Kingdom	+44 191 252 4850	+44 191 252 1054	c.l.j.frid@ncl.ac.uk
Simon P.R. Greenstreet	Fisheries Research Services Marine Laboratory P.O. Box 101 Victoria Road Aberdeen AB11 9DB United Kingdom	+44 1 224 295 417	+44 1 224 295 511	greenstreet@marlab.ac.uk
Mikko Heino	Institute of Marine Research Box 1870 Nordnes N-5817 Bergen Norway	+47 5523 6962	+47 5523 8687	mikko@imr.no
Louize Hill	DRM, IPIMAR Avenida de Brasilia PT-1449–006 Lisbon Portugal	+351 21 302 7062	+351 21 301 5948	lhill@ipimar.pt

Simon Jennings	CEFAS Lowestoft Laboratory Pakefield Road Lowestoft Suffolk NR33 0HT UK	+44 1502 513865	+441502 524363	s.jennings@cefas.co.uk
John Joyce	Marine Institute, Abbotstown Snugboro Road Dublin 15 Ireland	+353 1 822 8208	++353 1 82 5078	john.joyce@marine.ie
Ellen Kenchington	Bedford Institute Oceanography DFO P.O. Box 1035, Dartmouth, NS B2Y 4T3 Canada	+1 902 426 2030	+1 902 426 1862	kenchingtone@mar.dfo-mpo.gc.ca
Bob Mohn	Dept. of Fisheries & Oceans Bedford Institute of Oceanography P.O. Box 1006 Dartmouth, NS B2Y 4A2 Canada	+1 902 426 4592		mohnr@mar.dfo-mpo.gc.ca
Gerjan Piet	RIVO-DLO P.O. Box 68 NL-1970 AB IJmuiden Netherlands	+31 255 564 660		g-j.piet@rivo.wag-ur.nl
Jake Rice	Canadian Science Advisory Secretariat 200 Kent Street Ottawa, Ontario K1A OE6, Canada	+1 613 940 0288	+1 613 954 0807	ricej@dfo-mpo.gc.ca
Stuart I. Rogers	CEFAS Lowestoft Laboratory Lowestoft Suffolk NR33 0HT United Kingdom	+44 1 502 524 267	+44 1 502 524 464	s.i.rogers@cefas.co.uk
Sigmar Arnar Steingrimsson	Marine Research Institute Skulagata 4, P.O. Box 1390 121 Reykjavik Iceland	+354 522 0240	+354 562 3790	simmi@hafro.is
Mark L. Tasker	Joint Nature Conservation Committee Dunnet House 7, Thistle Place Aberdeen AB10 1UZ United Kingdom	+44 1224 655 701	+44 1224 621 488	mark.tasker@jncc.gov.uk

ANNEX 2: AGENDA

1) Continue the work started in 2001 to develop the scientific components needed for provision of scientific advice required by an EcoQO framework;

2) Quantify the relative role of fishing activity on dynamics of the marine ecosystem and nutrient turnover, in comparison with other comparable human activities such as marine disposal, and mineral extraction, to the extent possible;

3) Continue the work plan to test hypotheses about which components of the marine ecosystem are most sensitive to bottom fishing impacts;

4) In response to the EC DG Fish request for an "evaluation of the impact of current fishing practices on non-target species, … and suggestions for appropriate mitigating measures", investigate ways to use data products produced by the Study Group on Discard and By-catch Information for ecosystem management studies [contingent on discard and by-catch from SGDBFI being available for further analyses]. Where data are sufficient, evaluate the impact of fishing on non-target species. Identify species and fisheries where mitigative actions may be warranted and, in such cases, propose and justify alternative mitigation measures;

5) Drawing on material compiled by SGCOR, summarize all available information on the distribution of cold-water corals in the ICES area. Based on experience from the ICES area in particular, and more generally from cold waters of northern, southern, and deep-sea areas of the world, relate, to the extent possible, the information on the distribution of corals in the ICES area to threats from fishing activities and other potential disturbances [EC DG Fish];

6) Consider the report of the former Planning Group on Comparing the Structure of Marine Ecosystems in the ICES Area and specifically advise on the areas to be used in ecosystem comparisons and the meta-data available for such comparisons;

7) Propose a process to be able to summarize available information on the distribution of other sensitive habitats in the ICES area, and evaluate the adequacy of the information as a basis for scientific advice for an "evaluation of the impact of current fishing practices on … sensitive habitats, and suggestions for appropriate mitigating measures"; this should include the definition of criteria or standards for determining what is a "sensitive habitat";

8) Propose a process to be able to obtain information to develop advisory forms appropriate to the preservation of genetic diversity, beginning with the initiation of an evaluation of the advisory forms and management approaches that would be necessary and sufficient for the protection of genetic diversity of exploited stocks, and stocks suffering substantial mortality as by-catch;

9) Propose a process to be able to obtain information to consider "ecological dependence in management advice, firstly addressing the groups of species with the ecological linkages that are known with high reliability to have strong ecological linkages", including specification of the data requirements and models that would be required to provide the scientific basis for a response to that request. Propose a workplan and timetable for ICES to prepare itself for developing that scientific advice;

10) Review progress of activities initiated in 2001 by the Planning Group for a Workshop on [Top-down] Ecosystem Modelling;

11) Provide an assessment of the data on which the justification of the habitats in the OSPAR Priority List of Threatened and Endangered Species and habitats will be based; this assessment should be to ensure that the data used for producing the justification are sufficiently reliable and adequate to serve as a basis for conclusions that the habitats concerned can be identified, consistently with the Texel-Faial criteria, as requiring action in accordance with the OSPAR Strategy on the Protection and Conservation of the Ecosystems and Biological Diversity of the Maritime Area.