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REPORT OF THE

WORKING GROUP ON ECOSYSTEM EFFECTS OF FISHING ACTIVITIES

ICES Headquarters

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International Council for the Exploration of the Sea

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1 OPENING OF THE MEETING

The Working Group on Ecosystem Effects of Fishing Activities (WGECO) met at ICES Headquarters, Copenhagen, from 22 April to 3 May 2001. Attendance at the meeting comprised:

Jeremy Collie	USA
Niels Daan	Netherlands
Andrey Dolgov	Russia
Lars Føyn	Norway
Alain Fréchet	Canada
Chris Frid	UK (England and Wales)
Simon Greenstreet	UK (Scotland)
Sture Hansson	Sweden
Louize Hill	Portugal
Ronald Lanters	Netherlands
Robert Mohn	Canada
Gerjan Piet	Netherlands
Stefan Ragnarson	Iceland
Jake Rice (Chair)	Canada
Stuart Rogers	UK (England and Wales)
Gorka Sancho	Spain
Mark Tasker	UK (Scotland)

The meeting was timed intentionally to overlap with the meeting of the Working Group on Marine Mammal Population Dynamics and Habitats (WGMMPH) for the first week, and the Study Group on Ecosystem Assessment and Monitoring (SGEAM) for the final three days. This was to allow coordination of work on Terms of Reference for all three groups, to address requests for advice from OSPAR regarding Ecological Qualities and Ecological Quality Objectives for the North Sea. There was frequent exchange of members among the groups on a number of occasions, although occasional visitors from the other groups are not all listed as participants above, nor did they participate in approval of the meeting report as it was developed.

Terms of Reference for the WGECO meeting were discussed on the first day, 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, while avoiding redundancies with work being done by other Working Groups.

Terms of Reference for the meeting (C.Res. 2000/ACME09) were to:

a) consider the application of habitat classification and mapping (including GIS) to integrated environmental management incorporating fishing effects;

b) in response to the request from OSPAR [OSPAR 2001/2.2], working with the Working Group on Marine Mammal Population Dynamics and Habitats (WGMMPH) and SGEAM, and taking account of the reports and background documents of the Oslo Workshop on the Ecosystem Approach and the Scheveningen Workshop on EcoQOs,
   i) provide recommendations for appropriate Ecological Quality Objective indices for marine mammals, and suggestions for appropriate Ecological Quality Objectives for North Sea mammal populations,
   ii) prepare provisional estimates for the current levels, reference points, and targets for the Ecological Quality Objective indices identified in i);

c) in response to the request from OSPAR [OSPAR 2001/2.3], working with the Working Group on Seabird Ecology (WGSE) and SGEAM, and taking account of the reports and background documents of the Oslo Workshop on the Ecosystem Approach and the Scheveningen Workshop on EcoQOs,
   i) provide recommendations for appropriate Ecological Quality Objective indices for North Sea seabird populations, and suggestions for appropriate Ecological Quality Objectives for North Sea seabird populations,
   ii) prepare provisional estimates for the current levels, reference points, and targets for the Ecological Quality Objective indices identified in i);

d) implement the workplan outlined in Section 8.2 of the 1999 Report of the Working Group on Ecosystem Effects of Fishing Activities (ICES CM/2000:ACME:02), to the fullest extent possible, with the objective of further developing testable hypotheses for evaluating which components of the marine ecosystem are most vulnerable to trawl impacts;

e) based on previous considerations of community metrices and ecosystem reference points, provide recommendations on the development of EcoQOs for fish and benthic communities.

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 requirements. It also extends thanks to Dr Gubbay (UK) and Dr de Boer (Netherlands) for making drafts of their papers being prepared for the OSPAR EcoQ initiative available to the Working Group. These drafts allowed the Working Group to carry its consideration of EcoQs in a community framework much further.

2 EXECUTIVE SUMMARY

This year, the workload of WGECO was dominated by Terms of Reference arising from OSPAR requests for advice on EcoQs and EcoQOs. A great deal of time was spent by the ICES Professional Advisers and several WG Chairs before the meeting trying to ensure that the work of several WGs was complementary rather than redundant. Nonetheless, WGECO participants clearly had trouble seeing what value they would add to work already done by WGSE and WGMMPH, and how the work of SGEAM and WGECO would interact. WGECO ended up taking on the entire issue of EcoQs, EcoQOs, and indicators at the community scale for fish and benthos, and commenting only briefly on work done by other working groups.

In attacking the problem of community-level EcoQs, EcoQOs, and indicators, WGECO decided early to bring the same level of scientific rigour to the issue that has characterized its past work on, for example, impacts of trawl gears on benthic species and habitats. Even more than in the past, this makes the products of WGECO appear possibly less visionary or ambitious than treatments of the same issue by other groups. WGECO offers no apologies for this possible perception; in fact, it remains proud that it is capable of not lowering scientific standards as ecological problems become more complex. However, to avoid misunderstandings about what approach we took and why, **Section 5** is quite lengthy, and explains our framework and rationale in depth. We think this framework should be considered seriously as a suitable framework for the entire task of identifying EcoQs, EcoQOs, and indicators, at many scales.

Section 5.1 is our philosophy chapter. It presents the historical context in Section 5.1.1, focusing naturally on OSPAR's evolving interest in EcoQs, as well as past conclusions of WGECO with regard to ecosystem objectives and status indicators. Section 5.1.2 explores a number of terminological issues. These are potentially serious, because the EcoQ community is using a number of words and phrases that are also used in fisheries advice from ICES, but using them with different meanings. WGECO, naturally, cannot decide which usage should have universality (although the discrepancies should be resolved between fisheries and environmental quality interests at the earliest opportunity), but we clarify how each term will be used in this report. Section 5.1.3 jumps into conceptual issues, rather than just terminological ones. We try to clarify what is an EcoQ, an EcoQO, and an indicator, a point about which we all began with some confusion. Based on debates over wordings as the final report was being approved, we doubt that we succeeded fully in clarifying these distinctions, and if we remain confused, we expect that the larger scientific community is likely to be even more so as this initiative proceeds. We also discuss the role of Science in this initiative. **Read this section** because we feel that the messages here are very important. We are deeply concerned about the way objective science and partisan advocacy are being confused in many discussions of marine environmental quality and ICES needs to be a leader in this vital area. The rest of Section 5.1.3 goes into detail about exactly the approach it took to its work. In particular, consistent with its arguments about the Role of Science, it does not propose any EcoQOs, and argues that ICES as a whole should not. Selection of EcoQOs *necessarily* involves identifying states of ecosystem qualities and society desires. Although science considerations are germane to identifying states of EcoQs that are not compatible with conservation, science cannot say where society should *want* to be within the range of acceptable states. Finally, Section 5.1.4 waxes even more conceptual, pointing out lessons from single-species management and science advice that we feel ought to be very informative in the community and ecosystem contexts. We note failures and their probable causes more than successes and discuss governance issues. It took many iterations to gain consensus on the wording for this section, and some readers may feel that we are pushing the boundaries of ICES mandate and expertise. Nonetheless, there was a clear conclusion that systems of governance cannot be decoupled fully from the programmes they are intended to implement, and there needs to be at least serious discussion about what systems of governance would be necessary if progress is to be made on the whole notion of ecosystem management.

Section 5.2 commences the presentation of WGECO's work to identify community-scale EcoQs and indicators. The three main properties for which EcoQs and indicators are required are biological diversity, ecological functionality, and spatial integrity. Beneath each main property, WGECO identified a number of component properties and candidate indicators for each one. In the case of Biological Diversity and Ecological Functionality, the lists of component properties and candidate indicators were fairly long, and potential indicators were found to be very unevenly distributed among properties. For example, the number of possible indices of diversity runs to scores or hundreds, whereas there were few indicators for some of the functional energetics properties. Although the lists were far from exhaustive, WGECO was confident that at least examples for all main community and ecosystem features of Biological Diversity and Ecological Function were included on our total list of more than 50 candidate indicators.

Section 5.3 described the evaluation framework applied to the candidate list of indicators. Each indicator was scored qualitatively with regard to eight criteria, developed from OSPAR and other sources. WGECO considered that metrics 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.

Each candidate indicator was evaluated on each criterion by individuals or small groups, and the results combined. Of the 50+ indicators, for fish communities only 21 and for benthic communities only 14 survived the first cut, which dropped any indicator that was given a score of zero (not possible at all, or possible only after extensive additional research and monitoring) by all evaluators on one or more criteria. When any indicator with a modal score of zero on one or more criteria was dropped, there were 7 indicators for fish communities and only 1 for benthos. Of the seven for fish communities, three cover size spectra and two others species composition.

The resultant list of potentially sound community indicators does not include indicators for a large number of ecosystem properties of great interest to at least the science community. This is not a comforting result, particularly in light of the widespread interest in moving forward with “ecosystem management”. WGECO’s conclusion is that for many important properties, the science is just not there to support science-based advice and management decision-making. This gap should not be closed by lowering the standards for the science needed as a foundation for ecosystem management. Rather, there is a need for different, as well as more, community and ecosystem research. Section 5 concludes with a discussion of three particularly glaring gaps: functionally valid indicators for biodiversity, for three aspects of ecosystem functionality (well-being of constituent individuals, environmental forcing of biological processes, and tropho-dynamic relationships), and, especially, spatial integrity. It was particularly troubling to WGECO that, notwithstanding the universal recognition that population fragmentation, landscape ecology, meta-population dynamics, etc., are vital to community and ecosystem status, no one could propose even a single metric widely enough used to be a candidate indicator for EcoQs about Spatial Integrity.

In contrast to Section 5, **Section 6** is very brief. It considers the reports of WGSE and WGMMPH with regard to the overall approach taken, and also asks if there are community-scale indicators needed for seabirds and marine mammals, to complement the indicators proposed by the respective Working Groups. WGECO noted that the approaches taken by the two Working Groups differ greatly from each other, and from the approaches used here. It explores some of the possible consequences of the differences, but does not second-guess the choices of either group. With regard to community-scale EcoQs and indicators, WGECO noted that both groups did not rule out the possibility of developing community-based metrics, but neither did either group propose any. WGECO considered this issue and could not suggest alternative community- or ecosystem-scale properties that would be of any greater help in the management of human activities in the marine environment with reference to marine mammals and seabird populations than those suggested for single species.

Section 7 picks up the fish and benthos community indicators that passed the selection process described in Section 5. These are:

Fish communities

- Length frequency (%age composition by size class; slope of size spectrum);
- Mean length / weight of fish within specified limits;
- Presence of indicator / charismatic / sensitive species;
- Species abundance (k-dominance curves; species composition);
- Maximum length (weighted mean Lmax of community);
- Mean and distribution of "body condition".

Benthos communities

- Presence of indicator / charismatic species.

Informed by a particularly thorough review prepared in advance of the meeting, this section considers how each of the 10 threats identified by OSPAR would or would not be picked up by the community-scale indicators that survived our screening. WGECO concluded that several of the aspects of the fish community represented by different metrics appeared to be related and could be traced to one specific type of human activity; fishery induces size-specific mortality which changes the size-structure of the population. Therefore, the proposed metrics for the North Sea fish community are the average weight of individual fish and the average maximum length. From a conservation perspective, appropriate EcoQOs would move these metrics towards a larger proportion of large fish and would improve fisheries yields. Neither metric would discriminate between treatments which simply allowed individuals of exploited species to grow larger (and live longer – i.e., lower mortality), and treatments which changed the species composition towards a higher proportion of species with larger maximum possible weights and lengths (redistributing mortality across species, away from ones with greater maximum sizes).

With regard to benthic communities, WGECO considered carefully the implications of presence of charismatic or sensitive species as the only successful indicator. The presence of indicator or sensitive species cannot measure all the properties of benthic communities. Three other metrics measuring different properties of benthic communities scored quite highly using the framework. These metrics were biomass, k-dominance curves, and the presence of non-indigenous species. Adoption of these as metrics of benthic EcoQ may address some of the shortcomings of the application of "the presence of indicator or sensitive taxa". This section also discusses the potential for spatial-based indicators of water quality and habitat integrity for benthic EcoQs as an alternative. Possible merits and shortcomings of each approach are identified, and no recommendation is forthcoming with the information currently available.

This section also includes critiques of two other documents being prepared for the OSPAR EcoQ initiative: one on Benthos and one on Threatened and Declining Species. For Threatened and Declining Species, WGECO identified possible problems with both the selection criteria being used to identify such species, and the candidate list of species, and made some suggestions for ways forward on this topic. Nonetheless, WGECO endorsed a proposed overarching EcoQO based upon the "absence of threatened and declining species in the North Sea where the principal causes of threat and decline are linked to human activities". This would seem a useful suggestion, because the single metric might be the number of such species and the objective to reduce the number to zero. For the document on benthic species, WGECO noted that some proposed indicators are very similar to ones proposed here, whereas others were considered and rejected by this group. Some of the Diversity indices and physiological anomalies, in particular, were questioned on several grounds.

Other sections of the report do not address the OSPAR EcoQ requests. **Section 3** is a concise section, picking up a theme identified at the previous WGECO meeting. The section explores the application of habitat mapping (including GIS) to integrated environmental management incorporating fishing effects. The section first considers the requirements for a habitat mapping and classification system that meets the management needs, and then looks in detail at the EUNIS system with regard to meeting these requirements. It notes the particularly great importance of spatial scale when addressing habitat issues. Depending on the type of threat, one might need habitat classification and mapping information at the scale of square metres, or at the scale of hundreds to thousands of square kilometres. Following the review of the EUNIS system, an operational framework for it is proposed. Concern is expressed that the scale at which effective management will take place is at one or more square kilometres, but the habitat resolution required for such management in marine systems is at EUNIS levels 4 and 5, which we expect to be on the scale of a few metres. There is therefore an inherent mismatch between management needs and the ability of EUNIS to provide appropriate information at this scale. WGECO therefore recommends that EUNIS applies a higher degree of standardization to the habitats at levels 4 and 5. Several other operational recommendations are also made. However, with regard to WGECO interests, the main point is the inherent incompatibility of the current EUNIS spatial scales for bio-habitat information, and the management questions about effects of fishing and fishing gears.

Section 4 picks up addition work identified at our last meeting, to advance efforts at developing testable hypotheses for evaluating which components of the marine ecosystem are most vulnerable to trawl impacts, while avoiding the circularity inherent in much of the work we had reviewed in the past. The section first develops a list of characteristics and traits for which it was thought there could be sound theoretical, or common sense, grounds for being able to predict a clear directional response to variation in fishing impact. Next the behaviour of as many of these characteristics and traits in some real data sets has been tested. Both spatial and temporal analyses have been undertaken, looking for differences among areas differing in the level of fishing disturbance to which they have been subjected, and variation over time in areas where fishing impact has either increased or decreased. Three sets of data collected in different regions have been examined. In two data sets, the data available are used to rigorously test the various hypotheses. In the third data set, time series are looked at to determine whether the trends observed could give cause for concern in that area. The effect of fishing on life history characteristics within a particular species was not considered, for example, does the age at maturity of individuals within a fished cod population decline? This is a more complex problem which, although of great interest, was beyond the scope of the WG to address at this meeting.

The predicted responses to increased fishing disturbance were expressed as testable hypotheses: five at the scale of differential on species with different life-history traits, seven at the scale of changes in the mean or distribution of life-history traits across communities relative to fishing histories; two for ecosystem productivity, three for trophic structure; three for community diversity, and two for species' well-being.

The section presents an extensive summary of diverse analytical results on groundfish trawl survey data sets. As expected with such a diverse set of analyses, conducted in a limited amount of time, some hypotheses were confirmed, some were not, and many analyses were not definitive. Nonetheless, for at least some cases mean life-history characteristics can detect effects of trawling. Again these results seem to suggest, as emphasised by WGECO in the past, that the application of a suite of metrics provides more information than any single metric alone. Moreover, metrics seem most sensitive and informative when detecting initial impacts of fishing. Overall WGECO concluded that the results should be considered only as illustrations of the metrics. Even if all the analytical tools are working properly, the data are not sufficient to select among the proposed indices; none failed conspicuously nor did any excel. Further research should be conducted in three areas: including expansion to other sets of data; refinement of the metrics; and the development of a more methodical screening procedure. Much work remains in this important area, and some will be pursued intersessionally.

Sections 9 and 10 are the wrap-up sections. In **Food for Thought** a more complete framework is proposed for screening candidate indicators of ecosystem status or EcoQ. These ideas developed as the meeting grappled with our terms of reference. They did not coalesce in time to be retrospectively implemented in a second screening bout, and not all the necessary data would have been available, had the framework come together sooner. Nonetheless, it is recorded for future reference. The section also includes a discussion of the possible uses of CPUE as a management tool. Recent changes to fisheries management approaches and technological support may have changed some properties of CPUE. This section discusses some ideas of its new information content; ideas again to be explored further.

Section 9 is devoted to a discussion of the **future of WGECO**. For four meetings we have worked in a schizophrenic mode, partly advancing basic and applied knowledge through original and cooperative research, and partly as a review group collating existing knowledge and integrating it as support for ICES' advisory duties. We think that we do both jobs very well. However, progress on the first role is always frustratingly slow and incomplete, because there are always time deadlines for the second role. The section develops these concerns in detail, and presents options for consideration by ICES.

As a closing note, this meeting completed my three-meeting term as Chair of WGECO. As I travel I frequently hear very positive comments about the exceptionally high scientific standards that WGECO has brought to some very complex and highly public issues, as well as the breadth of issues tackled with that excellence. I cannot take credit for those achievements, because the credit belongs to all the participants who have worked so well and so hard through each meeting. However, I can, and do, take pride in them, and in the many friendships I have made or strengthened in my term as Chair. It has been one of the highlights of my career. I take pleasure knowing that the Working Group will still be in excellent hands when I step down. I look forward to working as a member of team for many meetings to come.

3 CONSIDER THE APPLICATION OF HABITAT MAPPING (INCLUDING GIS) TO INTEGRATED ENVIRONMENTAL MANAGEMENT INCORPORATING FISHING EFFECTS

3.1 Introduction

In addressing this Term of Reference, WGECO discussed the recent development of habitat classification systems and the way in which they may be used in the preparation of broad-scale mapping of the marine environment, and thereby contribute to the management of human activities. The discussion was based on the progress made by the Working Group on Marine Habitat Mapping (ICES, 2000a; Davies and Moss, 2000). Important features of a habitat classification system are identified and compared with what is currently available and potential methods for implementing environmental management using habitat maps are presented as an operational framework.

3.2 Habitat Classification and Mapping System Requirements

Human activities in the marine environment must be managed in such a way that habitat degradation and loss are minimised. This will not only ensure continued existence of ecologically important habitats, but also acknowledge that the protection of habitats is a prerequisite for the protection of associated species. Of the range of human activities being deployed at sea, fishing, aggregate extraction, dredging and disposal, coastal protection and land reclamation are known to affect marine benthic habitats. Pelagic environments are also under pressure from issues relating to water quality (eutrophication, pollution), harvesting and altered hydrography (coastal works, climate change and offshore structures). This section will focus on the information that is required for effective management of benthic habitats, outline the aspects involved in a broad-scale mapping programme, and describe the way in which this could contribute to the design of spatial management regimes.

Before addressing the issue of selection of habitats at risk and developing management measures, we first need a useful definition of habitats in terms of environmental management. The definition used in the EU Habitat Directive (European Communities, 1992) seems suitable in this respect: *"terrestrial or aquatic areas defined by geographic, abiotic and biotic features, whether entirely natural or semi-natural"*. This definition is pragmatic, allows for maximum flexibility and circumvents ongoing academic discussions. It also allows the identification of artificial reefs and shipwrecks (and other possible permanent man-made structures) as semi-natural habitats.

Another important consideration relates to the criteria for selecting marine habitats that are at risk or that need special attention to prevent further or future loss or degradation. Potential criteria for detecting such benthic habitats are, for instance:

a) the sensitivity and vulnerability of habitats to specific human impacts;

b) importance and specificity of their ecological functions;

c) their perceived intrinsic value (biodiversity);

d) the rarity of particular habitats.

WGECO has previously used the sensitivity to physical impacts of bottom gears to identify those benthic habitats under most threat (ICES, 2000b). Removal and destruction of physical and biogenic features were regarded as potentially the most damaging effect of bottom trawling. Also reduction in the complexity of habitats and alteration of small physical structures were regarded as a serious threat for the conservation of ecosystem integrity. Similar considerations apply to the effects of similar activities such as marine aggregate extraction and dredge spoil disposal (ICES, 2000c). Detailed reviews of the potential impacts of human activities are presented elsewhere (e.g., Jennings and Kaiser, 1998; Lindeboom and de Groot, 1998). The ecological functions of habitats offer a much wider scope for consideration. Habitats may serve as feeding areas, or nursery and breeding grounds, and all of these may be important at some stage in the life cycle of a species or groups of species. The intrinsic value of habitats can, for example, be determined by the presence of rare or charismatic species, or even based on human perception without supporting ecological data (although scientists have little to contribute in this case).

The range of management measures required will vary considerably between the range of human impacts under consideration, and the scale and range of habitat types. For example, the protection of the coral *Lophelia pertusa* will require detailed spatial information at a relatively small scale (metres), while the protection of seacaves and rocky reefs requires data on a larger scale (kilometres), and an interpretation of the precise extent of the habitat. As area-based management measures may seriously impact the conduct of human activities, the manager must be provided with accurate information on spatial distribution, and at the appropriate scale. However, the operational framework to be developed should allow for management actions at all scales.

3.3 EUNIS Classification System for Marine Habitats

3.3.1 Introduction

Recent ToRs for the Study Group on Marine Habitat Mapping include the review of developments in marine habitat mapping and specifically the preparation of habitat maps of the North Sea. WGECO considers that such developments are an essential precursor to the management of habitats and species on a broad scale, for a range of human activities which includes fishing effects. However, before habitat maps are prepared it is necessary to agree on a habitat classification scheme that provides relevant information about habitats, and applies a uniform set of criteria. The following section provides a brief outline of the classification system selected by the SGMHM, the European Nature Information System, EUNIS (ICES, 2000a), and an assessment of its utility within a framework of integrated environmental management.

The EUNIS hierarchical habitat classification has been developed on behalf of the European Environment Agency, and builds on earlier classifications for Palaearctic habitats and in Europe by CORINE. Ultimately, EUNIS should describe the complete range of habitats in the entire OSPAR area. Through the provision of a standard terminology and database, EUNIS intends to facilitate the dissemination of environmental data and to act as a repository for such information. Clearly, the selection of suitable organisms and scale of the habitats in which they occur are important to the habitat definition. The definition currently used is "plant and animal communities as the characterising elements of the biotic environment, with abiotic factors operating together at a similar scale". The main aims of EUNIS are to:

a) provide a common format for the description of all European marine, freshwater and terrestrial habitats;
b) be objective and clearly defined;
c) provide data in a relational database;
d) seek consensus amongst developers and users;
e) be comprehensive and operate at a number of hierarchical levels;
f) be flexible enough to evolve.

3.3.2 Characteristics

The scale proposed for the EUNIS habitat classification in the marine environment is that occupied by large invertebrates and vascular plants. It is considered that samples of between 1 m^2 and 100 m^2 are generally adequate to categorise habitats. At a smaller scale, and at a lower hierarchical level, micro-habitats which occupy less than 1 m^2 are included. At a larger scale, habitat complexes or frequently occurring mosaics of individual habitat types usually occupying about 10 ha, such as estuaries, mud flats, saltmarshes, etc., are used.

The main criteria, which are used in the classification to distinguish between successive hierarchical levels, are the ecological or biogeographical factors which determine the plant and animal communities. There are 10 major subdivisions at level 1, of which only "Marine habitats" and "Coastal habitats" apply to the marine environment. Level 1 marine habitats include, at level 2, all the littoral and subtidal marine environments and are subdivided into the following habitat types:

A1) Littoral and other hard substrata,
A2) Littoral sediments,
A3) Sublittoral rock and other hard substrata,
A4) Sublittoral sediments,
A5) Bathyal zone,
A6) Abyssal zone,
A7) Pelagic water column.

Of these level 2 habitats, the sublittoral rock and sediment classifications will contribute to the majority of the description of the NW European shelf habitat. They can be further subdivided to level 3, for example:

A3. Sublittoral rock and other hard substrata:

A3.1 Infralittoral rock very exposed to wave action and/or currents and tidal streams,
A3.2 Infralittoral rock moderately exposed to wave action and/or currents and tidal streams,

A3.3 Infralittoral rock sheltered from wave action and currents and tidal streams,

A3.4 Caves, overhangs and surge gullies in the infralittoral zone,

A3.5 Circalittoral rock very exposed to wave action or currents and tidal streams,

A3.6 Circalittoral rock moderately exposed to wave action or currents and tidal streams,

A3.7 Circalittoral rock sheltered from wave action and currents including tidal streams,

A3.8 Caves, overhangs and surge gullies in the circalittoral zone,

A3.9 Deep circalittoral rock habitats,

A3.A Vents and seeps in sublittoral rock.

A4. Sublittoral sediments:

A4.1 Sublittoral mobile cobbles, gravels and coarse sands,

A4.2 Sublittoral sands and muddy sands,

A4.3 Sublittoral muds,

A4.4 Sublittoral mixed sediments,

A4.5 Shallow-water sediments dominated by angiosperms (other than *Posidonia*),

A4.6 *Posidonia* beds,

A4.7 Deep circalittoral sediment habitats,

A4.8 Seeps and vents in sublittoral sediments.

A more comprehensive subdivision of these level 3 habitats exists within the EUNIS classification, and this reflects the greater degree of knowledge that exists for these relatively well-studied coastal and offshore demersal environments. Importantly for the study of fishing effects on demersal habitats, it is at this stage in the hierarchy, level 4, that biota begins to be used increasingly to characterise habitat types. So, for example, the level 3 classification “A4.2 Sublittoral sands and muddy sands” is subdivided into level 4 as follows:

A4.2 Sublittoral sands and muddy sands:

A4.2/B-IGS.FaS(p)	Animal communities in fully marine shallow clean sands,
A4.2/M-III.2.1.	Biocenosis of fine sands in very shallow waters,
A4.2/M-III.2.2.	Biocenosis of well-sorted fine sands,
A4.2/B-IGS.EstGS	Animal communities in variable or reduced salinity shallow clean sands,
A4.2/H-02.05.02	Baltic brackish water sublittoral biocenoses of sands influenced by varying salinity,
A4.2/B-IMS.FaMS	Animal communities in fully marine shallow-water muddy sands,
A4.2/O-	Animal communities in variable or reduced salinity muddy sands,
A4.28	Animal communities in circalittoral muddy sands,
A4.2/M-IV.2.1.	Biocenosis of the muddy detritic bottom.

Further subdivision of the level 4 habitat such as “Animal communities in fully marine shallow-water muddy sands” into level 5 habitats is as follows:

A4.2/B-IMS.FaMS	**Animal communities in fully marine shallow-water muddy sands:**
A4.2/B-IMS.FaMS.EcorEns	*Echinocardium cordatum* and *Ensis* sp. in lower shore or shallow sublittoral muddy fine sand.
A4.2/B-IMS.FaMS.SpiSpi	*Spio filicornis* and *Spiophanes bombyx* infralittoral clean or muddy sand.
A4.2/B-IMS.FaMS.MacAbr	*Macoma balthica* and *Abra alba* in infralittoral muddy sand or mud.
A4.2/B-IMS.FaMS.Cap	*Capitella capitata* in enriched sublittoral muddy sediments.

While level 3 classifications are almost complete for the shelf seas demersal habitats, there are some incomplete classifications at the functional groups identified at level 4 and especially at level 5.

It is generally at levels 4 and 5 that habitats are described in terms of both their physical and biological characteristics. It is this information which is most appropriate when evaluating the impact of human activities on habitats.

3.4 Application to the North Sea

Interpreting the response of a habitat to human impact usually depends on the magnitude and duration of the impact. In the EUNIS classification, the “marine environment” is only classified at level 1 as a subset of a range of other mainly terrestrial habitats, and this has prevented the inclusion of the necessary range of perturbed and unimpacted marine habitats at a high level. In terrestrial systems, for example, habitats such as “regularly or recently cultivated agricultural and domestic habitats” and “constructed, industrial and other artificial habitats” have been included at level 1 (ICES, 2000a). Many equivalent marine examples exist. A comparable classification of marine habitats would have been valuable in EUNIS, mainly because the spatial extent of impacted areas is better known, and more attention has been given within the scientific community to the response of species and habitats at such impacted sites.

The identification of such habitats could also include man-made structures such as reefs or licensed sites in which extraction and disposal of sediments have taken place. These support specific habitats with different responses to perturbation. Inclusion of unimpacted areas dominated by fragile structural biota would also be valuable, rather than link such fauna within a broad abiotic habitat description, which will support a range of less sensitive species.

It is important that EUNIS identifies marine habitats that are either sensitive to human impacts or that are locally unusual and/or vulnerable, and that such areas can be detected at a relatively high level in the classification, in order for its use as a tool to conserve these habitats. Previously, WGECO (ICES, 2000b) ranked the potential effects of bottom trawls on habitats and species according to the sensitivity of the habitat. That classification was based on existing evidence of trawl impact studies reviewed at the meeting. Trawl impact studies have identified the kinds of habitats that are at greatest risk from anthropogenic impacts, but the large-scale distribution of these habitats is less well known. For EUNIS to make a full contribution to this process, it is necessary that descriptions of habitats refer both to the biotic and the physical environment, at a relatively high level in the classification. This should be taken into account in plans to develop North Sea maps at levels 3 and 4, because otherwise such maps are expected to be of limited practical value in the management context. There is therefore urgent need for better baseline data on the distribution of both fauna and abiotic characteristics of the habitat, and the inclusion of both aspects in habitat maps. The EUNIS structure has not identified clear spatial scales at each level of habitat discrimination, and this may complicate the process of effective environmental management. The scale at which effective management will take place is at one or more square kilometres, and the habitat resolution required for this is at EUNIS levels 4 and 5, which we expect to be on the scale of a few metres. Management schemes in the future may be implemented at the scale of ICES rectangles down to a few square kilometres at specific features. However, our knowledge of human impacts and appropriate management responses have been gained from studies of habitat types carried out on small spatial scales, equivalent to habitats described at levels 4 and 5.

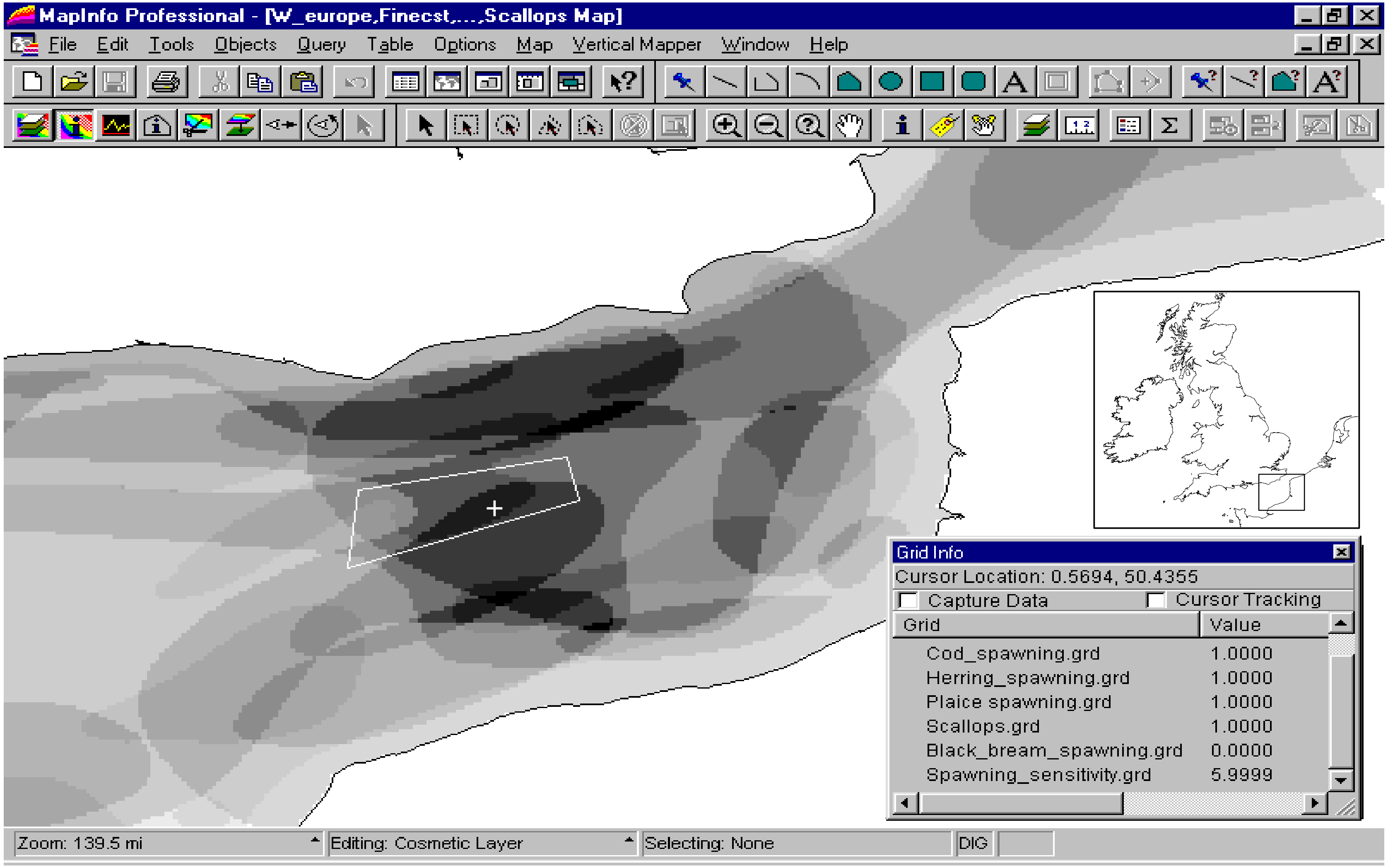
In recent years, there have been technological advances in acoustic mapping techniques, such as in the use of multibeam and side-scan sonars to map the topography of the seabed (e.g., Kenny *et al.*, 2000). Furthermore, these techniques can be used to discriminate between major sediment and habitat types, although this generally requires groundtruthing by seabed sampling. Acoustic techniques can at most only discriminate between major habitat types, and other techniques such as remotely operated vehicles, underwater photography and grab sampling are required to investigate benthic communities on smaller scales, which may be appropriate at level 5. Very few studies have used both approaches within the same area to map simultaneously the distribution of fauna and habitats (e.g., Todd *et al.,* 2000).

3.5 Proposed Operational Framework

3.5.1 Introduction

A number of practical considerations need to be taken into account when developing a management framework that incorporates these habitat classifications. Foremost amongst these is the need to ensure that map outputs are compatible with modern geographic information software (GIS), and are consistent with corresponding spatial data describing the distribution of human activities. Thus habitat maps should allow the matching of point source impacts, such as oil and gas platforms, as well as those that are measured across broader ranges, such as beam trawl effort at the scale of an ICES rectangle. This capability will be necessary to establish accurate geographic correspondence between habitats and human activities, and ensure that effects can be evaluated for habitat types at different levels of resolution. For example, a human activity may broadly impact a large-scale habitat (such as sublittoral sediments, Level 2) but the real environmental effects might be better measurable at lower-scale levels (i.e., changes in biological communities at Levels 4 and 5).

Figure 3.5.2.1. A cumulative sensitivity map of the eastern English Channel, showing shaded areas of sensitivity from low (light grey) to high (dark grey) derived from the number of fish species spawning in the area. The white polygon represents a hypothetical area of human impact and the cursor (white cross) is used to retrieve information at a chosen geographical coordinate. At this location a "Grid Info" dialogue box gives information about the species which contribute to the total sensitivity, and other data are stored in the attribute table.



An issue of great concern is the practical limitation on any programme to prepare maps of extensive areas of the North Sea, even at level 2 of the existing habitat classification scheme. It must be recognised that the cost of new acoustic surveys and groundtruthing for this task will be prohibitively expensive. The alternative solution, where existing data sets are collated into a consistent format, has other problems. These issues have already been discussed by WGMHM (ICES, 2001), and are clearly a cause for concern. Finally, it is clear that mapping effort should be dynamically and periodically revised to account for changes of biotic communities. This will ensure that the process has a time frame that makes it acceptable as an environmental monitoring tool.

3.5.2 The application of habitat mapping in an integrated environmental management context

Mapping systems such as GIS provide efficient electronic data storage for multiple data sets. There is a need to begin using these spatial analytical tools to support advice on the effects of human activities on broad scales. This section describes examples of using GIS in this context.

The outline of the area of interest (nursery ground, spawning area, reef) must be digitised and stored as a series of joined x and y coordinates (i.e., in vector format). This digitising process creates electronic maps of geographically referenced polygons, normally from paper charts. Any information concerning the polygon, such as the habitat name, the EUNIS level, the species it represents, seasonality, etc., is stored in an attribute table. This table can also store data describing the importance or "value" of the site. This is useful in more advanced analyses because it provides a method for comparing areas of different importance, and allows different levels of sensitivity to be allocated to different species.

Each digital map can be viewed in isolation. However, one of the benefits of GIS is that a number of different features can be overlayed and viewed together, thereby allowing the impact to be assessed in terms of all relevant parameters of the marine environment. Before these polygons can be overlaid and viewed, however, they must be converted to a raster or grid-file format. This process converts a vector plot to a grid by dividing the area up into many small cells. Each cell in these new grid-files has a numeric value taken from the attribute table. In its simplest form, this is 0 or 1 depending on whether the species is present or absent from that cell.

A derivative map is produced from the combination of several such grid-files. In the example presented here, a derivative map describing the spawning distribution of fish species has been produced, and each cell describes the combined total of species which spawn in that cell. The example shown in Figure 3.5.2.1 is a preliminary output from this process for the Eastern Channel. This is used only as one example of a suitable technique and maps of habitat and the extent of human activity can be used in the same way. The areas of greatest intensity of shading represent those locations that support the most fish and shellfish species. The importance of overlaying and combining these maps using GIS is that the same cell in each "layer" can be queried to show which species contribute to the total fish density at that position.

The benefit of the GIS approach is that the user can choose which maps to include, perhaps based on the vulnerability to specific activities, or the relative importance of species in terms of their economic or conservation value. The degree of overlap between habitat maps and areas of human impact can be used to assess the potential impact of those activities. The GIS process allows the user to calculate the area of the impacted habitat as a proportion of the total. Knowing how much of a particular resource is impacted by human activity is the first step towards evaluating effects.

3.5.3 Inferring habitat patterns using spatial analysis

Conventional broad-scale marine sampling of benthic species or assemblages requires samples to be collected at well-spaced intervals, so there is no information for the unsampled areas between sites. However, data sets with 100 % coverage of large areas of offshore waters are available for the abiotic factors which structure these demersal communities, such as seabed surface sediment type, water current speed at the seabed, water temperature, salinity, etc. If sufficiently robust associations can be made between bottom-dwelling fauna and the physical structure of the seabed they are found on, then these 100 % coverage digitised sediment charts could provide the spatial dimension that is currently lacking for the fauna. Future development of this approach can use the spatial statistical functions that are available in GIS to predict the spatial extent of benthic assemblages/species. Maps created in this way will still need to be validated using existing and new biological survey data, but they provide a tractable solution to the problem of mapping extensive marine benthic resources.

3.6 Conclusions and Recommendations

- The scale at which effective management will take place is at one or more square kilometres, but the habitat resolution required for this is at EUNIS levels 4 and 5, which we expect to be on the scale of a few metres. There is therefore an inherrent mismatch between management needs and the ability of EUNIS to provide appropriate information at this scale. WGECO therefore recommends that EUNIS apply a higher degree of standardization to the habitats at levels 4 and 5.
- It is important that habitat maps are based on a logical classification of the marine environment. This is available as the EUNIS habitat classification scheme, and further efforts to populate the lower hierarchical levels should be encouraged.
- WGECO recommends that future developments of the EUNIS classification scheme take into account habitats influenced by human activity.
- WGECO also recommends that, where possible, habitat maps are prepared using descriptors of biological communities as well as the physical substrate. The biological information will be required by environmental managers in order to effectively manage activities which have explicit spatial dimensions.
- Effective management of many types of impact requires spatially explicit information on both the extent of the threat, and the habitats threatened. Habitat maps provide this in an accessible form. WGECO urges that more use be made of GIS to assist management decisions, and suggests appropriate methodologies to facilitate movement in this direction.

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4 TESTABLE ECOLOGICAL HYPOTHESES ABOUT FISHING EFFECTS

4.1 Further Develop Testable Hypotheses for Evaluating which Components of the Marine Ecosystem are Most Vulnerable to Trawl Impacts

This term of reference required us to:

"implement the workplan outlined in Section 8.2 of the 1999 Report of the Working Group on Ecosystem Effects of Fishing Activities (ICES, 2000), to the fullest extent possible, with the objective of further developing testable hypotheses for evaluating which components of the marine ecosystem are most vulnerable to trawl impacts." Vulnerability is interpreted here as being sensitivity to fishing disturbance, without necessarily being currently disturbed.

In this report, WGECO was concerned with the apparent circularity in logic prevalent in the identification of vulnerable species. Rarely were such species proposed *a priori* on the basis of theoretical expectations. More commonly, vulnerable species were identified on the basis of their observed response to changes in fishing pressure. This circularity leads to uncertainty in the interpretation of data, and contributes to a degree of confusion regarding the value of particular species, or particular traits, as indicators of the impact of fishing on the ecosystem.

WGECO suggested that, "A more useful approach might be to ask ourselves the question whether we are able to classify species on the basis of life history characteristics in a ranking order for vulnerability and then test the hypothesis that those species which have been classified as the most vulnerable have actually shown a decline in response to fishing and vice versa." Such a classification should take place before any analysis of the data. This has been taken a step further and the question—"can mean-values of these characteristics, weighted by species abundance, be used to monitor the effects of fishing on the entire fish community?"—has been asked.

To follow up this recommendation, a list of characteristics and traits for which it was thought there could be sound theoretical, or common sense, grounds for being able to predict a clear directional response to variation in fishing impact was suggested. In carrying this forward, WGECO drew initially from the list of potential candidates and ideas presented in the previous report. A few further possibilities have also been considered. Next the behaviour of as many of these characteristics and traits in some real data sets has been tested. Both spatial and temporal analyses have been undertaken, looking for differences among areas differing in the level of fishing disturbance to which they have been subjected, and variation over time in areas where fishing impact has either increased or decreased. Three sets of data collected in different regions have been examined. In two data sets, the data available are used to rigorously test the various hypotheses. In the third data set, time series are looked at to determine whether the trends observed could give cause for concern in that area.

The analyses have been restricted to the potential impacts of fishing on fish species, and on trends in the mean characteristic value determined for the entire (or components of the) fish community. Furthermore, these analyses are preliminary, and the results did not receive full scrutiny of the WG. There are plans to revisit this work and continue it (see Section 9). In addition, there is no reason why a similar approach could not be adopted for other components of the marine ecosystem, such as birds, marine mammals, invertebrates and benthic communities. The effect of fishing on life history characteristics within a particular species was not considered, for example, does the age at maturity of individuals within a fished cod population decline? This is another complex problem which, although of great interest, was beyond the scope of the WG to address in the time available.

4.2 Specific Hypotheses Regarding the Impact of Fishing on the Characteristics and Traits of Fish Communities

In this section many traits and characteristics of both species and fish communities are considered, *a priori* hypotheses are stated, and other initiatives are set out.

4.2.1 Specific hypotheses

When trying to characterise species based on life history characteristics there is extensive literature that distinguishes K-strategists from r-strategists on theoretical grounds and how their relative abundance in a community depends on the stability of their environment. This division is debatable and here the method is used without judgement that the validity of this approach has been established definitively. Rather, life history reasoning is commonly used in interpreting the results of studies of impacts of fishing. The purpose here is to ask, "Given the theoretical framework, can predictions of fishing effects be made and tested on a more *a priori* basis?"

According to this paradigm, K-strategists are adapted to living in stable and predictable environments and have greater competitive ability. They have longer life-spans, larger body size, reproduce later, produce few young, and are more likely to exhibit parental care. In contrast, r-strategists live in unpredictable or disturbed environments. They are small organisms with short life-spans, early reproduction and high fecundity. All gradations between the two extremes are possible and in practice it may be difficult to characterise a particular species as belonging to either strategy. Therefore the approach chosen here was to use the values of life history parameters to indicate a species' position on the r/K continuum.

Fishing pressure increases mortality in all species. When this increase in mortality is placed in the context of life history theory it may have several consequences. Life history traits that change as a response to fishing will change in the same direction for all species, but the rates of change in the life history parameters should differ in important and informative ways. For a specific level of fishing mortality, populations of species with K-selected traits will decline faster than species with r-selected traits. Furthermore the life history characteristics will change faster for species at the K-end of the continuum. Thus the predicted responses by individual species to increased fishing disturbance are expressed below as testable hypotheses relative to what would be expected for r-selected species:

- Species with large ultimate body length (L_{max} or L_{inf}) should decline;
- Species with slow growth rates (e.g., k from the von Bertalanffy equation) should decline;
- Species with older age at maturity (A_{mat}) should decline;
- Species with longer length at maturity (L_{mat}) should decline;
- Species with a low fecundity and lower life-time reproductive output should decline.

For a given level of fishing mortality, at the community scale the percentage of the community composed of species with K-selected traits will decline. Because of responses predicted for individual species, the response of each of these characteristics calculated across the assemblage as a whole is predictable. The community average character values, weighted by species abundance, should respond to an increase in fishing disturbance as follows:

• L_{max}	Decrease
• L_{inf}	Decrease
• Growth Rate	Increase
• Fecundity	Increase
• Life-time reproductive output	Increase
• A_{mat}	Decrease
• L_{mat}	Decrease

These changes in growth rate and fecundity should affect the productivity of the fish assemblage. Thus:

- The overall production to biomass (P/B) ratio of the fish community should be higher in more intensively fished areas, and it should increase as fishing disturbance increases.

The trophic level at which fish feed is strongly size dependent; larger fish in the community tend to be piscivores, smaller fish are planktivores and/or benthivores. With the decline of larger fish in more heavily fished areas, or as fishing in an area increases, the trophic structure of the community should change (e.g., Pauly *et al.,* 2001).

- Species that feed at higher trophic level will be more sensitive and should decline as fishing effort increases, or have a lower abundance in heavily fished areas.
- The average trophic level of the fish community should decline as fishing intensity increases, and be lower in more heavily fished regions.

Because of an increase in the amount of damaged and killed benthic organisms left lying on the seabed as a consequence of demersal fishing, species best able to utilise this resource are likely to increase in abundance.

- The proportion of fish that can be considered scavengers should have increased in intensively fished areas.

Species with obligate habitat requirements should decline in abundance when such habitat is lost as a consequence of fishing activity.

- Species that depend on a three-dimensional habitat (e.g., a fragile biogenic habitat) should decline in abundance and have a lower abundance in areas where habitat is altered by increased levels of trawling.

WGECO 2000 suggested that species richness should decline more in intensively fished areas than in less disturbed areas. This can be tested in two ways:

- Spatially, species richness should be lower in areas of high fishing intensity;
- Temporally, species richness should decline in areas where fishing intensity is increasing.

Many factors could confuse the response of species diversity to changes in fishing levels. Huston's (1994) dynamic equilibrium model suggests that the response of species diversity to disturbance is dependent upon local productivity.

- Species diversity should decline in response to increased fishing disturbance in areas of low productivity. In areas of high productivity, increased fishing could cause species diversity to increase.

Fish also pass through the meshes of the gear and can become damaged in the process, which could increase mortality and susceptibility to disease.

- The prevalence of fish showing sub-lethal effects (scarring, scale loss, external lesions, etc.) in intensively fished areas should be higher compared with fish in relatively undisturbed regions.
- Species which are particularly sensitive to the effects of scale loss, etc., are likely to decline in abundance as fishing intensity increases, and to have lower abundance in areas of high fishing activity, relative to insensitive species.

4.2.2 Other initiatives

In Section 5.3.3 of this report, concern is expressed that there may be attributes of the distribution of species or groups of species that have not been well explored. This section attempts to develop some spatial metrics and apply them in a provisional way to fisheries survey data. In Atlantic Canada two metrics of distribution are commonly reported in single species assessments: the area covered by a species and an index of concentration, which is the area containing the densest portion of the resource (Branton and Black, 2000).

The proposed metrics are applicable to single species and are aggregated into community or group indices. It is not clear, or perhaps even likely, that the aggregated indices will be more valuable than those for specific single species. As the work is exploratory it is offered as a stimulus to further work as opposed to a definitive or proscriptive study.

There is an implicit hypothesis that the distribution of animals may affect their viability and further that some species will be more sensitive to displacement than others. Further if species are perturbed, their community may also be affected. For example, if they are scattered too widely, they may be subject to higher predation or compromised recruitment. Conversely, if they are concentrated into a small area, they could suffer increased fishing mortality per unit effort (Paloheimo-Dickie effect (Paloheimo and Dickie, 1964)). Another consideration is displacement from traditional spawning areas. It is further assumed that fishing activity (or pollution) could affect these distributions. At this time hypotheses about which species or species groups will be most affected have not been posed.

4.3 Approach

The analysis performed by WGECO had two purposes:

Firstly, to use the most comprehensive data available to test a set of specific hypotheses with the purpose of identifying those characteristics and traits of fish species and communities that might be most useful as metrics of trawling impact. This required three basic types of information:

1) Trawl survey data providing information on the species' abundance in samples of fish. These data must extend over sufficient time so as to have substantial contrast in fishing events if the hypotheses being tested involve the evaluation of temporal trends. Alternatively, if the tests involve spatial comparisons, a reasonably large and dynamic geographic ranged is required.
2) Information on species characteristics or traits, for example, age at maturity, or habitat requirements of the species caught in the trawl surveys. Such information is required for a sufficient number of species so as to ensure that a reasonably large fraction of the total number of individuals caught in each sample are included.

3) Information on the variation in fishing effort, over time and/or space, is necessary for directly testing hypotheses. Ideally the temporal and/or spatial extent of the data should match that/those of the groundfish survey data.

Secondly, to explore the potential use of these metrics as indicators of impending ecological problems for managers. Thus temporal and spatial trends in the abundance of potentially sensitive species were explored, as well as trends in the metrics calculated for the entire fish community in an area where fishing effort data were not available. The reasoning behind this analysis was to explore whether the fish community concerned was affected by fishing activity in the area.

4.4 Analysis of the Data Sets

4.4.1 Northwest North Sea (Scottish August groundfish surveys)

In this section we use Scottish August Groundfish Survey (SAGFS) data, international and Scottish fishing effort data, and information on life history characteristics of the species encountered in the survey data, to test some specific hypotheses derived from the theoretical expectations presented in Section 4.2. A primary objective of the section is to identify which, if any, of the life history parameters examined might hold potential as a metric of fishing-induced change in the fish community, and the particular circumstances where the greatest insight might be gained. The data presented cover 75 ICES statistical rectangles located in the northwestern North Sea where data coverage is most reliable (Figure 4.4.1.1).

4.4.1.1 Species characteristics

Information regarding four life history characteristics (L_{inf}, Growth Rates, A_{mat}, L_{mat}) was available for 32 of the species (Jennings *et al.*, 1998, 1999a) encountered in the SAGFS, listed in Table 4.4.1.1.1. The L_{inf} and Growth Rate were the parameter values determined from the von Bertolanffy growth equation calculated for each species. The von Bertolanffy parameter is not strictly a rate value, but here is used as an index equivalent to growth rates. A_{mat} and L_{mat} values were determined by observation, either from recent survey data or with recourse to the literature. These 32 species accounted for over 99 % of the individuals sampled by the SAGFS in each of the 75 statistical rectangles. No life history characteristic information was available for the remaining 24 species included in the database. These species were among the rarest sampled, and combined they represented less than 1 % of the total number of individuals sampled in any rectangle. Their influence on the mean value of each characteristic could only have been negligible. For the purposes of this analysis, therefore, abundance data for these species were excluded. In the final temporal analysis, the proportion of the sampled fish assemblage in any time-period/"treatment" cell never dropped below 98.5 %.

4.4.1.2 Effort

International otter trawl, beam trawl, and Seine net fishing effort (hours fished) for the period 1990 to 1995 were available from the database compiled as part of the EC "Monitoring Biodiversity…" project (Jennings *et al.*, 1999b, 2000). Average annual effort values were calculated to provide estimates of the 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.

The possibility that the life history composition of the groundfish assemblage was affected not only by the absolute amount of fishing effort in any statistical rectangle, but also by recent trends, was also considered. The international fishing effort database covered only the years 1990 to 1995 and so does not provide much of a time series. The Scottish fishing effort database extends further back in time (Greenstreet *et al.*, 1999b). Furthermore, Scottish vessels landing in Scotland account for most of the fishing effort in this part of the North Sea. Indices of annual rates of change in otter trawl, beam trawl, and Seine net effort were therefore determined for each of the 75 statistical rectangles using the Scottish data. Effort data for Seine net and 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 average annual rates of change for each gear in each rectangle over the 25-year period. Beam trawling is a relatively recent phenomenon in the northwestern North Sea, and effort data for this gear were only recorded from 1984 onwards. The same approach described above was adopted, except that the start point five-year period was 1984 to 1989, and the divisor was 5, thus providing an annual rate of change in beam trawl use index over a ten-year period.

Figure 4.4.1.1. Area of the North Sea covered by the data sets analysed in this section.

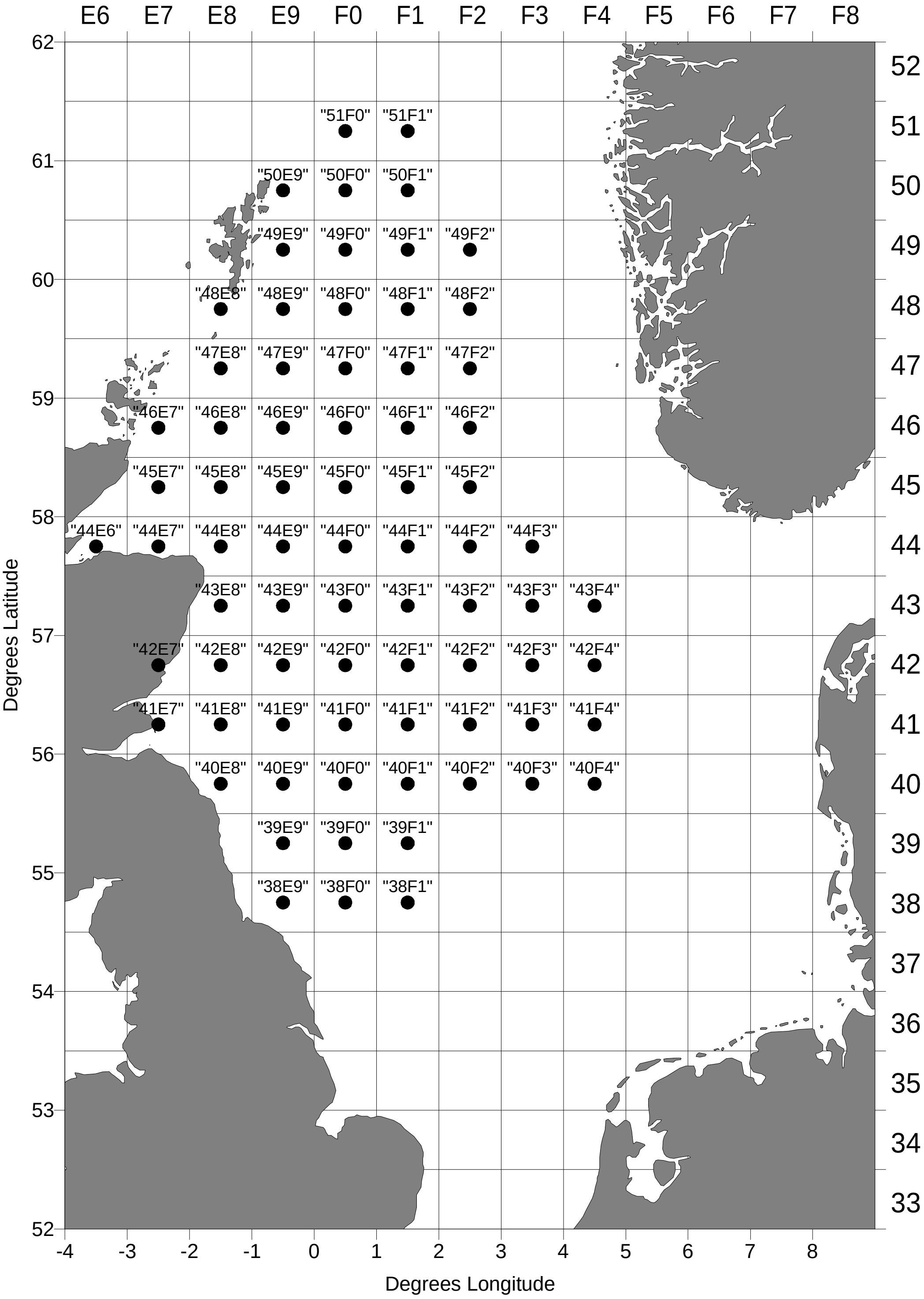


Table 4.4.1.1.1. List of species encountered in the SAGFS for which life history character information was available.

Angler	*Lophius piscatorius*
Bull rout	*Myoxocephalus scorpius*
Catfish	*Anarhichas lupus*
Cod	*Gadus morhua*
Common dab	*Limanda limanda*
Cuckoo ray	*Raja naevus*
Dover sole	*Solea solea*
Dragonet	*Callionymus lyra*
Four-bearded rockling	*Enchelyopus cimbrius*
Grey gurnard	*Eutrigla gurnardus*
Haddock	*Melanogrammus aeglefinus*
Hake	*Merluccius merluccius*
Halibut	*Hippoglossus hippoglossus*
Hooknose	*Agonus cataphractus*
Lemon sole	*Microstomus kitt*
Lesser spotted dogfish	*Scyliorhinus canicula*
Long rough dab	*Hippoglossoides platessoides*
Megrim	*Lepidorhombus whiffiagonis*
Norway pout	*Trisopterus esmarki*
Plaice	*Pleuronectes platessa*
Poor cod	*Trisopterus minutus*
Saithe	*Pollachius virens*
Skate	*Raja batis*
Spotted ray	*Raja montagui*
Spurdog	*Squalus acanthias*
Starry ray	*Raja radiata*
Thornback ray	*Raja clavata*
Three-bearded rockling	*Gaidropsarus vulgaris*
Torsk	*Brosme brosme*
Turbot	*Scophthalmus maximus*
Whiting	*Merlangius merlangus*
Witch	*Glyptocephalus cynoglossus*

4.4.1.3 Survey (catch) data

Groundfish survey data collected as part of the Scottish August Groundfish Surveys were examined. Trawl species abundance data were extracted for 75 ICES statistical rectangles in the northwestern North Sea covering a period of 14 years from 1983 to 1996. Up to four trawl samples were then excluded as necessary in order to reduce the number of samples to 10 in all rectangles. For one rectangle, data for the years 1983, 1985, 1987 and 1995 were missing. In reducing the number of trawls to 10 in the remaining rectangles, samples from these years were selected and deleted at random as required. All ten trawl samples in each rectangle were then pooled to provide a single aggregated sample for each rectangle. The samples for each rectangle were thus standardised as far as possible, given the type of sampling involved. All trawl samples were collected by the FRV "Scotia (II)", using a 48-foot Aberdeen Otter Trawl, towed for one hour (Greenstreet and Hall, 1996; Greenstreet *et al*., 1999a). The number of trawl samples per rectangle was the same for all rectangles, thus avoiding any sample-size dependency issues.

Species abundance data were converted to the number of individuals with particular characteristic values, and the mean value for each characteristic for each rectangle was computed. Data were available only for groundfish species likely to be well sampled by the gear. Pelagic species and other species not well sampled by the 48-ft Aberdeen otter trawl, such as herring, sprat and sandeels, were all excluded from the data set. The results therefore only apply to the demersal groundfish community occupying the area.

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

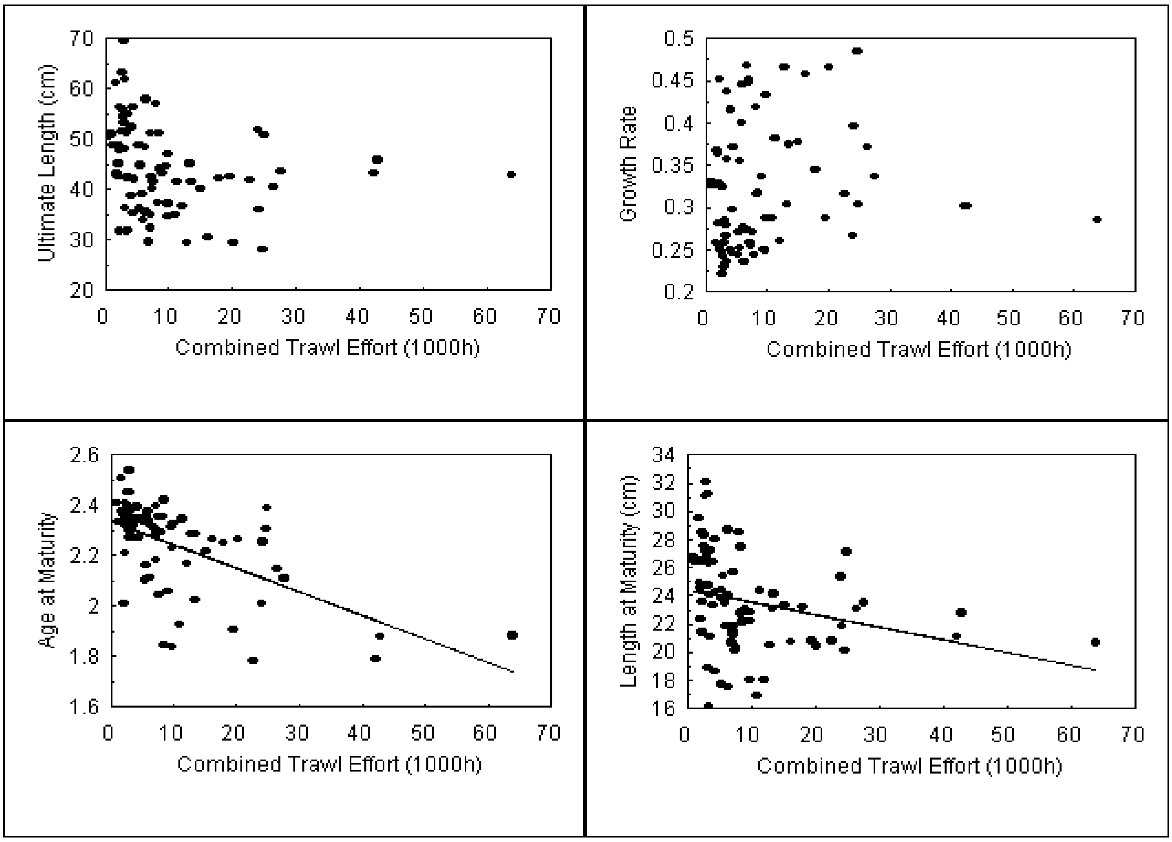
4.4.1.4 Analysis and results

This section is hypothesis driven. A series of specific hypotheses are presented, all of which are related to or derived from the theoretical discussion presented in Section 4.2. The data are then analysed so as to refute, or otherwise, each hypothesis.

Hypothesis: Groundfish assemblage Growth Rates should be positively correlated, and L_{inf}, L_{mat} and A_{mat} should be negatively correlated in space, with fishing effort.

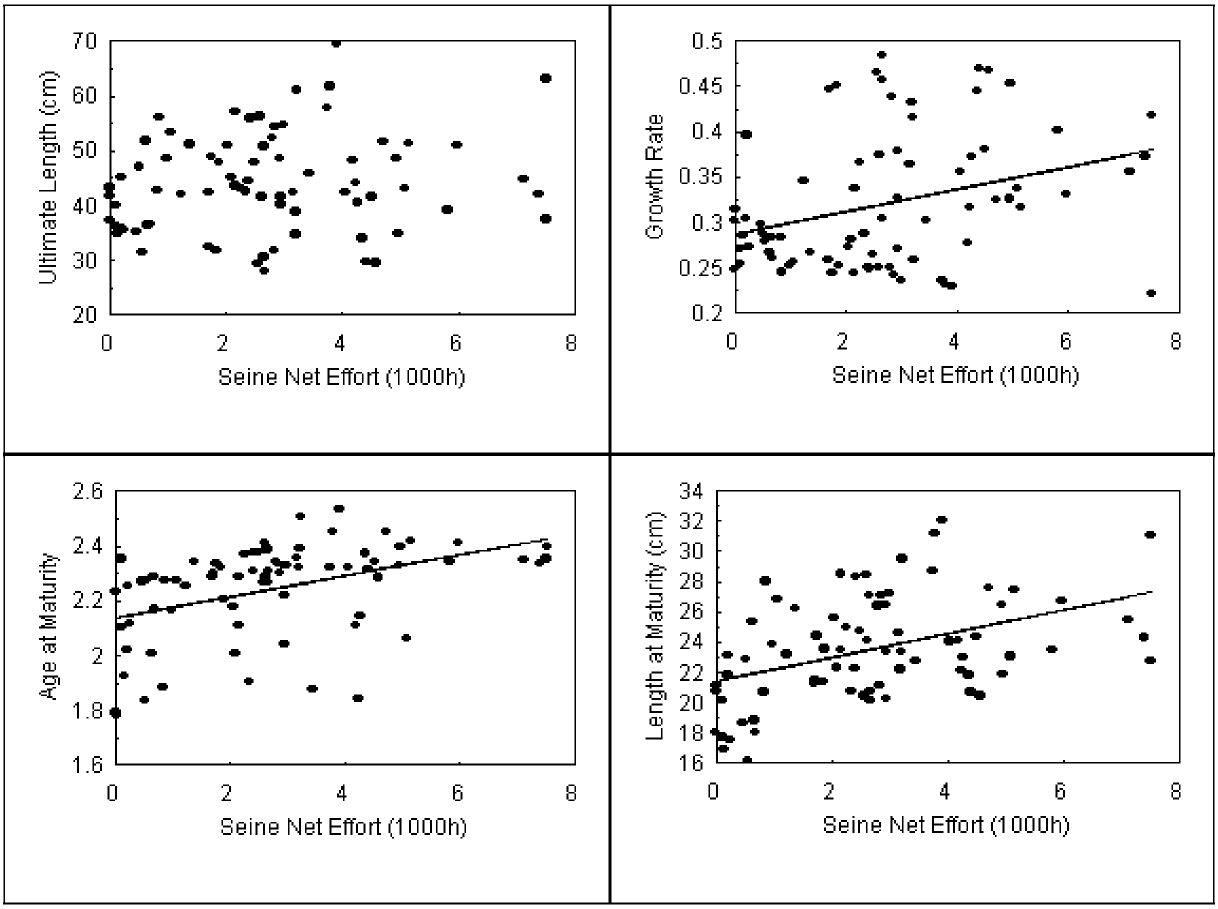
Life history charactistic data were plotted against international effort data. Relationships between assemblage average life-history characteristics and otter trawl and beam trawl effort in each of the 75 statistical rectangles were similar. Also, since otter trawl effort exceeded beam trawl effort by a factor of six, data for both gears were aggregated. The effects of the two gears combined on each of the life history characteristics were then examined (Figure 4.4.1.4.1). Correlation coefficients were computed and used as a guide to identify life history characteristics that could potentially be useful as metrics indicative of an effect of fishing on groundfish communities. All four life history characteristics responded to increased trawling effort in a manner predicted by our hypothesis, however, only the correlations for A_{mat} and L_{mat} were significant at the 5 % level. Furthermore, some caution is required in interpreting the significance of any of the correlations presented in this section, since the degrees of freedom applied take no account of the fact that these are essentially spatial analyses. Because of the strong possibility of spatial auto-correlation between many of the data points used, rendering them not truly independent of each other, the actual number of degrees of freedom is likely to be less than, in this case, 73.

Figure 4.4.1.4.1. Relationship between combined otter and beam trawl effort in 75 ICES statistical rectangles and the average and $Length_{Infinity}$, Growth Rate, $Age_{Maturity}$ and $Length_{Maturity}$ determined for 32 species making up >99 % of the total number of individuals sampled in each rectangle.



By the early 1990s, use of Seine net gear in this part of the North Sea had declined from the high levels characteristic of the 1960s. Nevertheless, the relationship between spatial variation in Seine net effort and the life history characters of the groundfish assemblage in each statistical rectangle was also explored (Figure 4.4.1.4.2). Three characters were correlated at the 5 % level of significance: Growth Rate, A_{mat} and L_{mat}. However, only the relationship for Growth Rate was in the direction predicted by our hypothesis. Seine net used to be the predominant type of gear used in a large part of the northwestern North Sea. Over recent decades this gear has largely been replaced by otter trawls. It is possible that the relationships displayed in Figure 4.4.1.4.2 have been influenced by this change in fishing practice, such that the relationships between any character and Seine net use have been affected by the impact of otter trawling in the same rectangles.

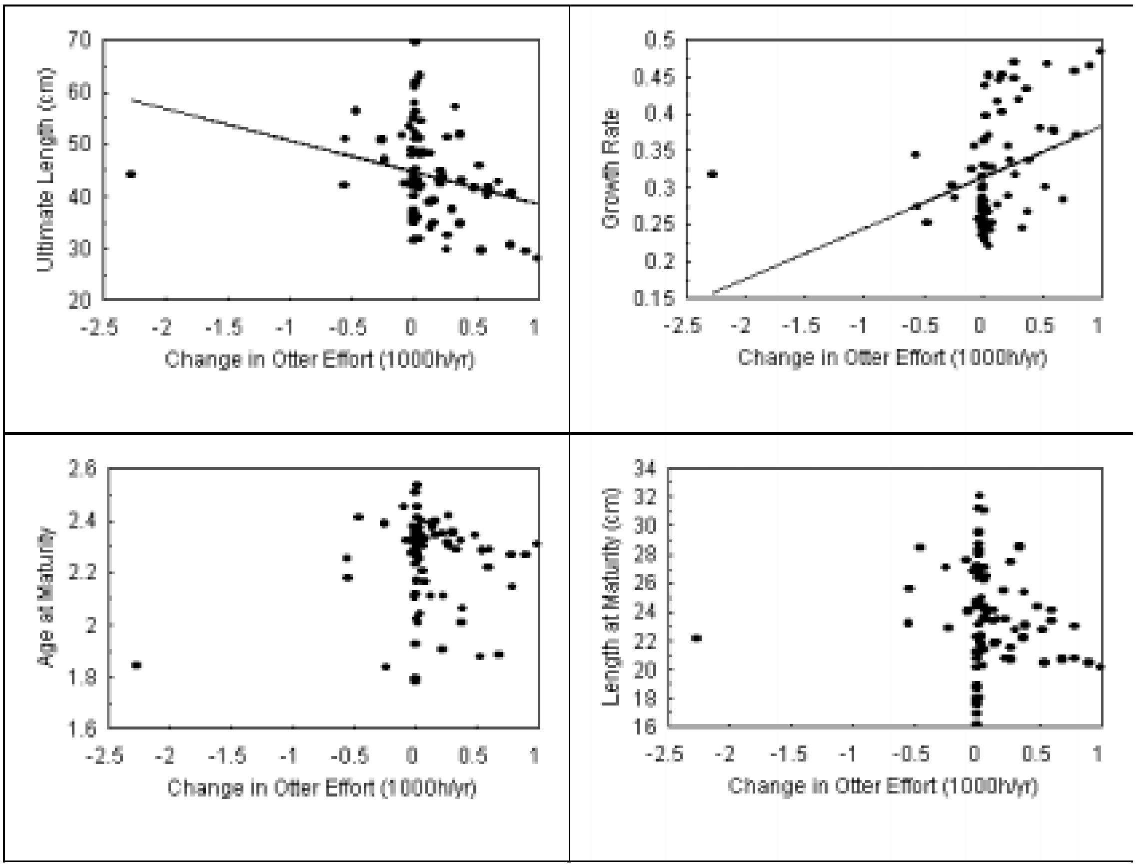
Figure 4.4.1.4.2. Relationship between Seine net effort in 75 ICES statistical rectangles and the average and $Length_{Infinity}$, Growth Rate, $Age_{Maturity}$ and $Length_{Maturity}$ determined for 32 species making up >99 % of the total number of individuals sampled in each rectangle.



Hypothesis: Groundfish assemblage Growth Rates should be positively correlated, and L_{inf}, L_{mat} and A_{mat} should be negatively correlated in space with rates of change in fishing effort over recent years.

Relationships between mean assemblage life history characteristics and annual rates of change in fishing effort in each rectangle were examined (Figure 4.4.1.4.3). The correlations for the L_{inf} and Growth Rate were significant at the 5 % level, whilst those for A_{mat} and L_{mat} were not. These two sets of results raise the possibility that assemblage mean L_{mat} and A_{mat} might provide indicators of the effect of absolute levels of fishing effort on the life history composition of the groundfish communities, while mean L_{inf} and Growth Rate could reflect recent changes in fishing effort. However, a small number of points have very high leverage in these calculations, so patterns must be viewed with extra caution.

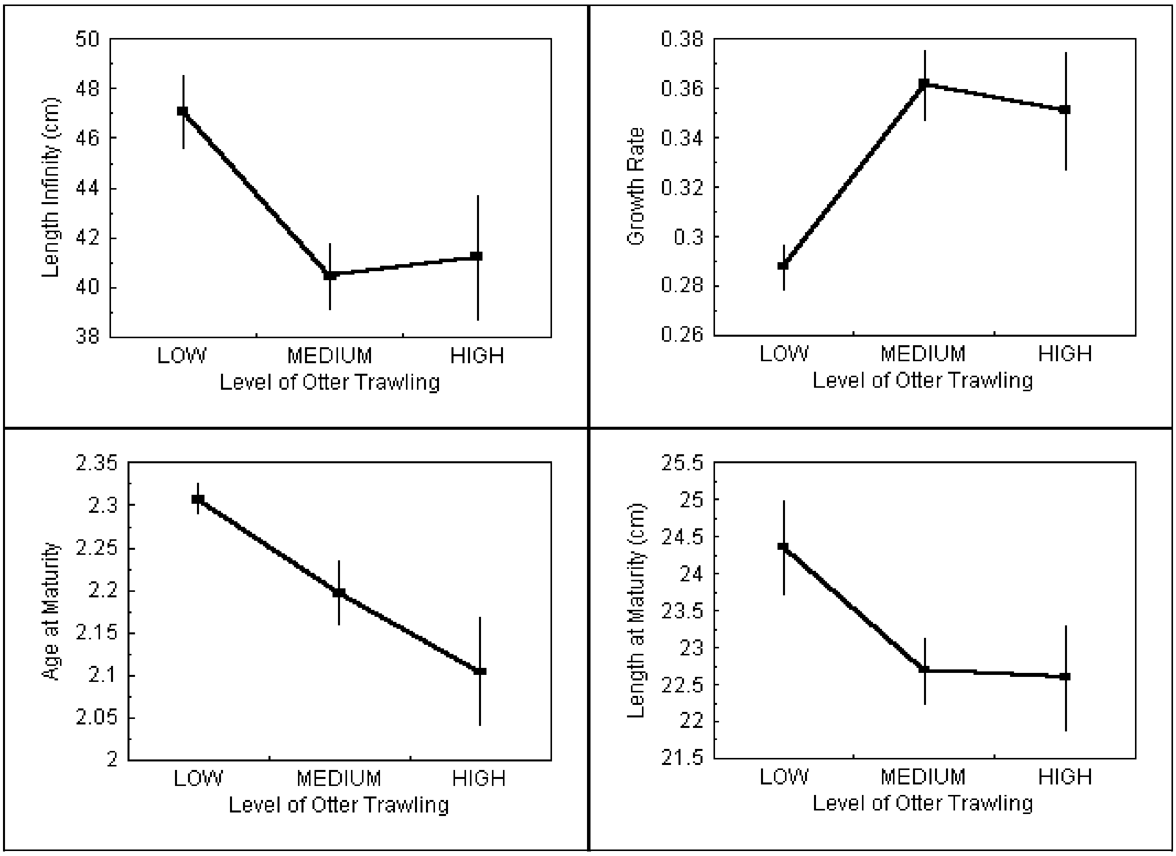
Figure 4.4.1.4.3. Relationship between annual average rate of change in otter trawl effort in 75 ICES statistical rectangles over the period 1970–1994 and the average and $Length_{Infinity}$, Growth Rate, $Age_{Maturity}$ and $Length_{Maturity}$ determined for 32 species making up >99 % of the total number of individuals sampled in each rectangle.



Hypothesis: Groundfish assemblage Growth Rates should be higher, and L_{inf}, L_{mat} and A_{mat} should be lower in areas with higher fishing effort.

Levels of international otter trawl fishing effort ranged from 645 hr yr^{-1} to 63,794 hr yr^{-1} across the 75 ICES statistical rectangles. The rectangles were sorted into three groups varying in the intensity to which they had been fished during the early 1990s: a low-intensity group of 40 rectangles where effort varied from 0 to 4,999 hr yr^{-1}; a medium-intensity group of 25 rectangles in which effort varied from 5,000 to 19,999 hr yr^{-1}; and a heavily fished group of 10 rectangles in which effort exceeded 20,000 hr yr^{-1}. The mean, and standard error of the mean, of each life history characteristic was determined for each group of rectangles (Figure 4.4.1.4.4). Differences, tested using one-way ANOVA, were found to be significant at the 1 % level for all four characteristics. In each case, the trend was consistent with the hypothesis.

Figure 4.4.1.4.4. Variation in the mean (± 1 S.E.) $Length_{Infinity}$, Growth Rate, $Age_{Maturity}$ and $Length_{Maturity}$ determined for 32 species making up >99 % of the total number of individuals sampled in each rectangle calculated for groups of rectangles varying in the level of otter trawl effort to which they were subjected.

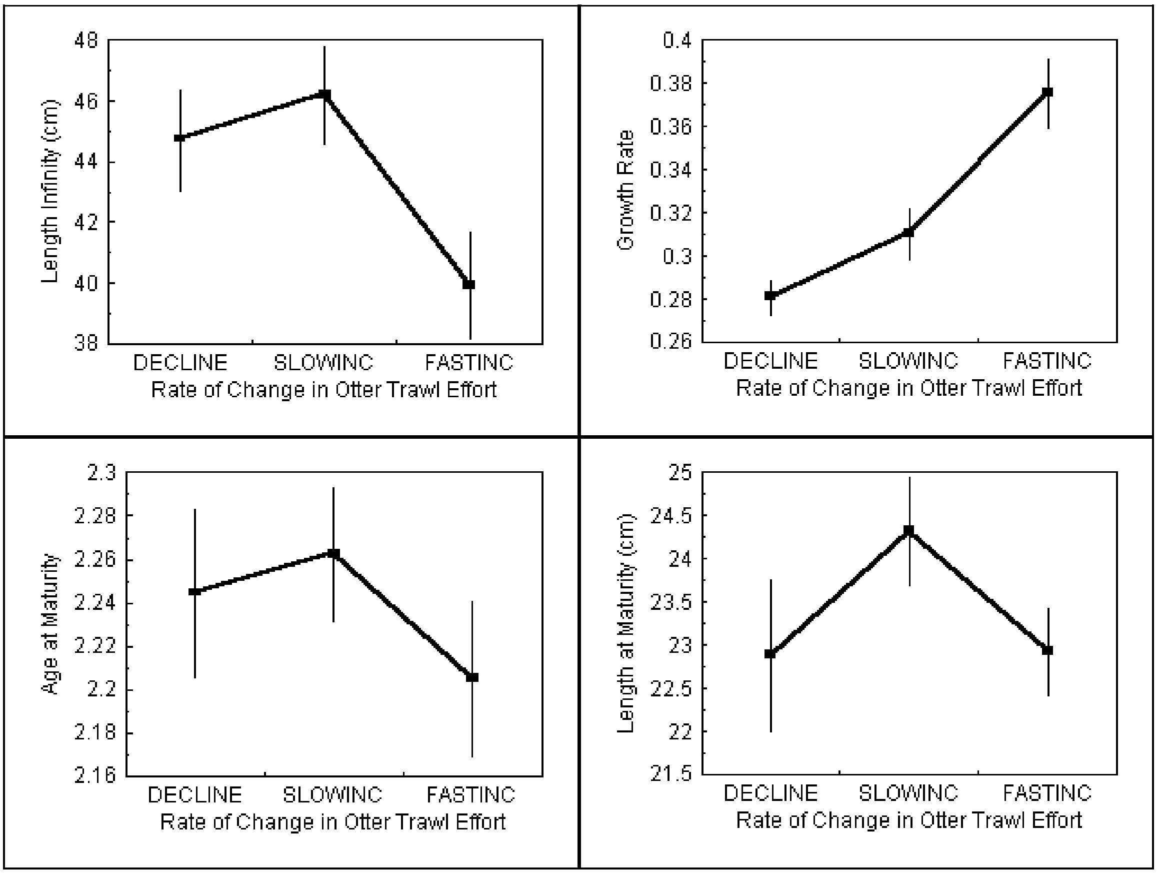


L_{inf}, Growth Rates and L_{mat} appeared to be the most sensitive characteristics, differentiating most between low and medium fishing intensity groups of rectangles. Beyond a certain (threshold?) level of perturbation, variation in these characteristics tended to level out. This raises the possibility that, as metrics, these three characteristics may be relatively insensitive in already disturbed areas. On the other hand, A_{mat} continued to decrease strongly as otter trawl effort increased from low, through medium, to high levels.

Hypothesis: Groundfish assemblage Growth Rates should be higher, and L_{inf}, L_{mat} and A_{mat} should be lower in areas where fishing effort is increasing at the greatest rate.

Annual rates of change in otter trawl effort varied from the extreme outlier of –2,268 hr yr^{-1} to 991 hr yr^{-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 hr yr^{-1}; and a group of 21 rectangles where effort was increasing rapidly, between 200 and 991 hr yr^{-1}. The mean, and standard error of the mean, of each life history characteristic was determined for each group of rectangles (Figure 4.4.1.4.5). Differences, tested using one-way ANOVA, were found to be significant at the 1 % level for all four characteristics. Only the Growth Rate behaved entirely as anticipated by the hypothesis. For the three remaining characteristics, the highest mean parameter values were observed on the rectangles with slow rates of increase in otter trawl effort.

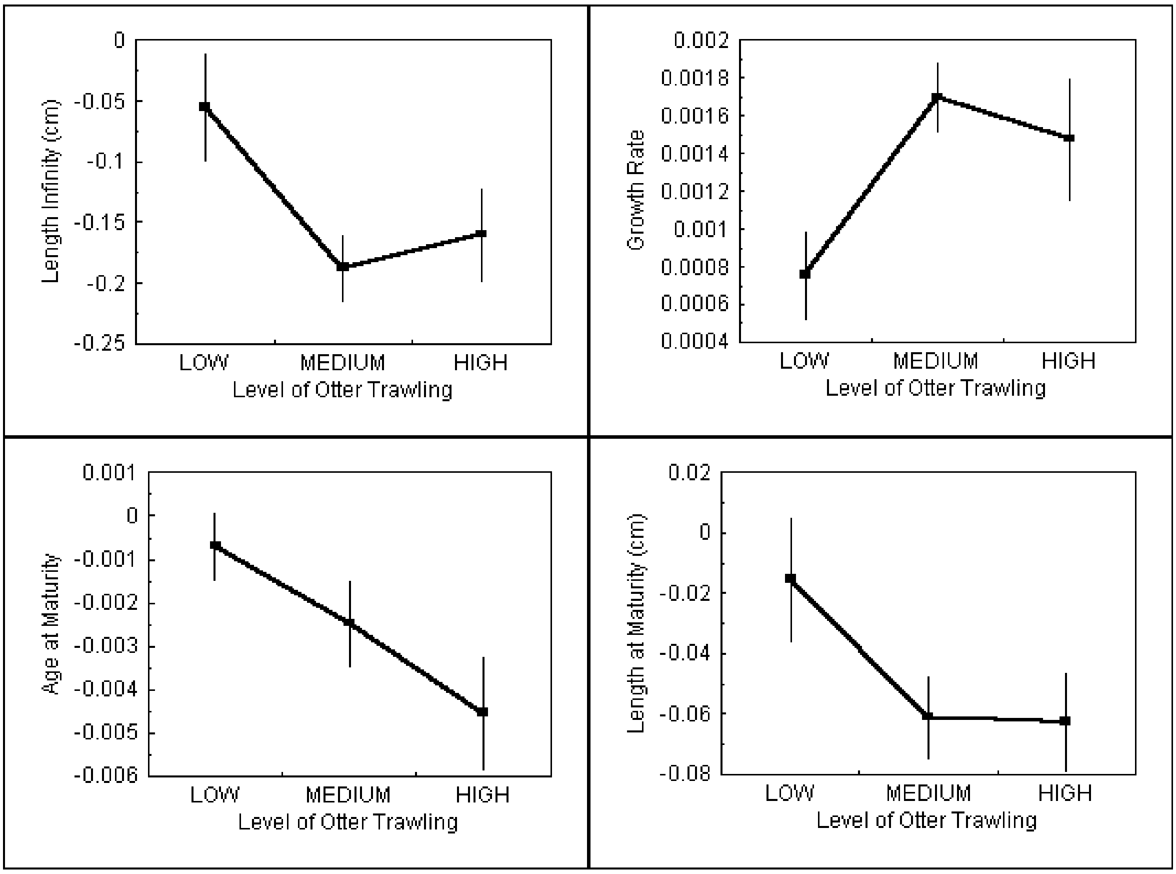
Figure 4.4.1.4.5. Variation in the mean (± 1 S.E.) $Length_{Infinity}$, Growth Rate, $Age_{Maturity}$ and $Length_{Maturity}$ determined for 32 species making up >99 % of the total number of individuals sampled in each rectangle in groups of rectangles with different annual rates of change in otter trawl effort.



Hypothesis: Long-term temporal variation in groundfish assemblage Growth Rates should show steeper positive trends, and L_{inf}, L_{mat} and A_{mat} steeper negative trends, in areas where fishing effort is higher, and in areas where recent trends in fishing effort have shown the greatest rates of increase.

This analysis used the full time series of available groundfish survey data, from 1925 to 1996, to explore the long-term behaviour of community mean life history characteristics in areas of varying fishing intensity. The rectangles were grouped into the same three treatment levels of international otter trawl effort during the early 1990s, and for recent trends in Scottish otter trawl effort over the period 1970 to 1994. As before, abundance-weighted mean character values for the groundfish community were determined for each time/treatment cell. 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 (Figures 4.4.1.4.6 and 4.4.1.4.7).

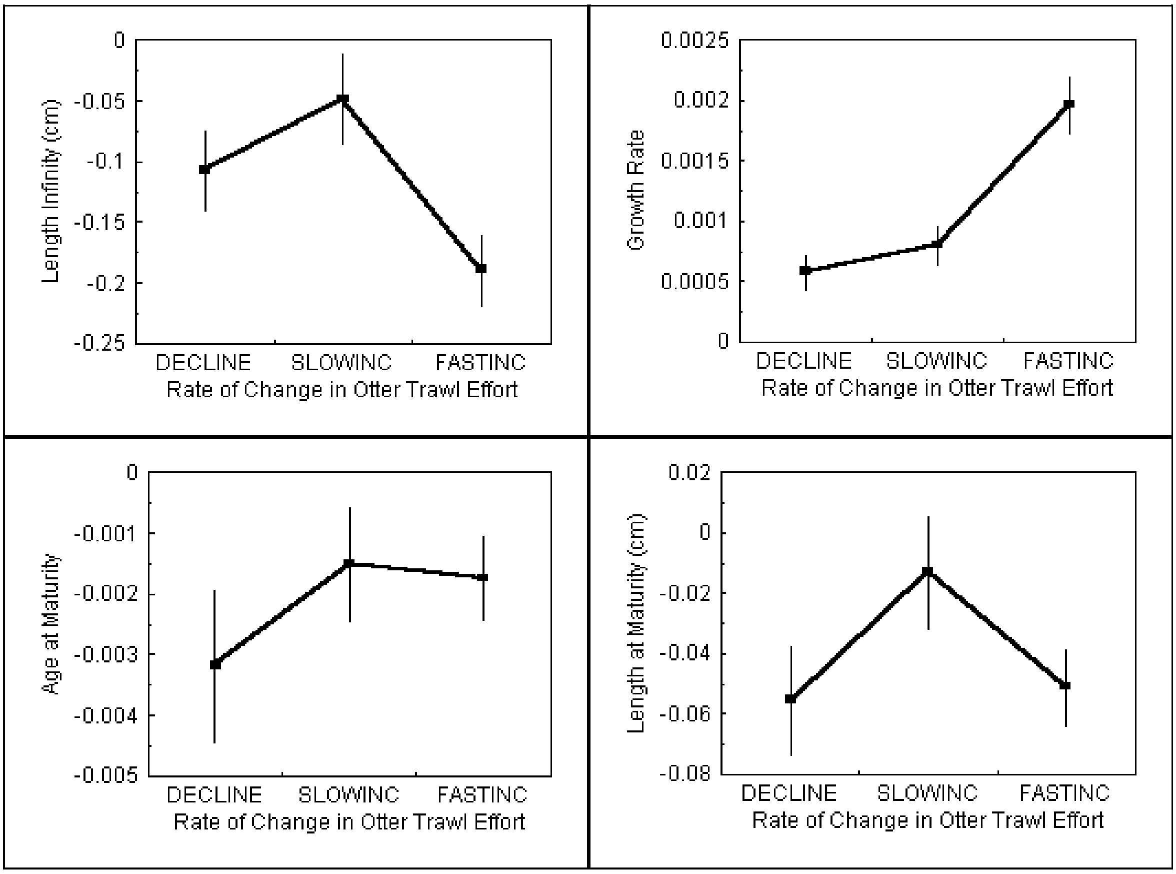
Figure 4.4.1.4.6. Variation in the regression coefficients (± 1 S.E. of the coefficient) for the slopes of $Length_{Infinity}$, Growth Rate, $Age_{Maturity}$ and $Length_{Maturity}$ over the time period 1925 to 1995 in rectangles with different mean levels of annual international otter trawl effort over the period 1990 to 1995.



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 (Figure 4.4.1.4.6). Of interest again was the fact that L_{inf}, Growth Rate and L_{mat} all failed to differentiate between medium and high levels of fishing effort. Again, this suggests that these parameters may be able to distinguish between fished and unfished areas, but once an area is impacted, they may be relatively insensitive to further perturbation. A_{mat}, however, showed increasingly steep long-term declines as otter trawl effort increased from medium to high levels of otter trawl activity. This analysis therefore again suggests 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.

The data presented in Figure 4.4.1.4.6 are also helpful in aiding our interpretation of Figure 4.4.1.4.4 as they suggest that the current community-averaged life history parameters (e.g., Figure 4.4.1.4.4) are the result of long-term changes from some earlier common, presumably near pristine, state.

Figure 4.4.1.4.7. Variation in the regression coefficients (± 1 S.E. of the coefficient) for the slopes of $Length_{Infinity}$, Growth Rate, $Age_{Maturity}$ and $Length_{Maturity}$ over the time period 1925 to 1995 in rectangles with different annual rates of change in Scottish otter trawl effort over the period 1970 to 1994.

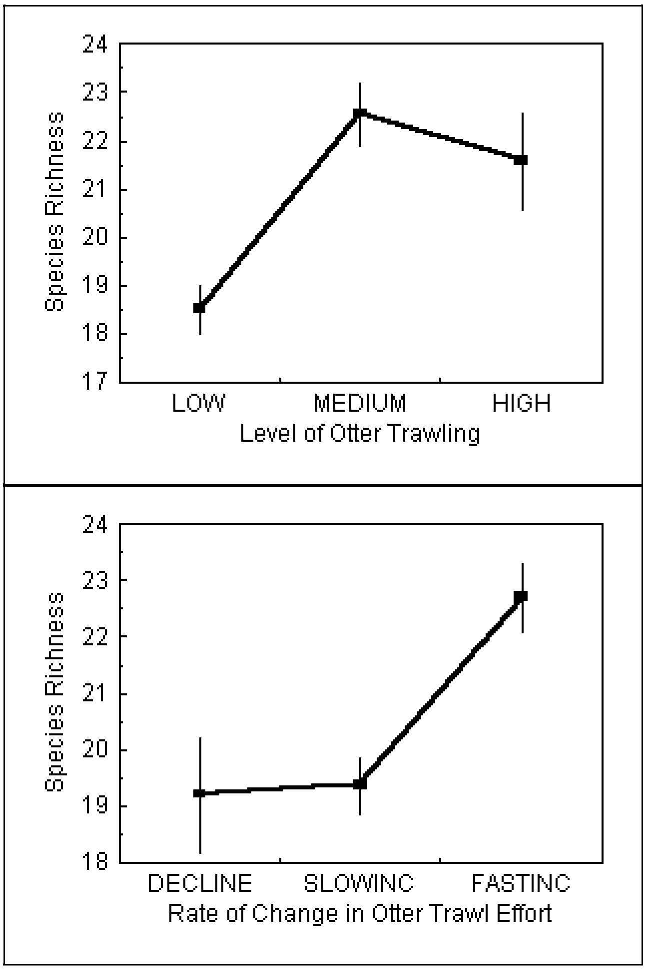


As with Figure 4.4.1.4.5, the interpretation of Figure 4.4.1.4.7 is more problematic. For example, Figure 4.4.1.4.7 suggests that the greatest long-term rates of decline in both A_{mat} and L_{mat} occurred in rectangles where otter trawl activity has actually declined over the period 1970 to 1994. This clearly contravenes the hypothesis. L_{mat} does show steeper long-term declines in rectangles where otter trawling has increased most rapidly over the period 1970 to 1994. A_{mat} fails even to do this. Variation in L_{inf} and Growth Rate, however, both support the hypothesis. The long-term decline in L_{inf} and long-term increase in Growth Rate are both steepest in the rectangles where otter trawling has increased most rapidly over the period 1970 to 1994.

Hypothesis: Species richness should be lower in areas where current levels of fishing effort are highest, and in areas where recent trends in fishing effort have shown the greatest increase.

Species richness was determined for each of the statistical rectangles from simple counts of all the different species recorded in each rectangle. Mean species richness was determined for three groups of rectangles with low (0 to 4,999 hr yr^{-1}), medium (5000 to 19,999 hr yr^{-1}), and high levels (>20,000 hr yr^{-1}) of fishing intensity. The same approach was adopted for examining the effect of trends in fishing effort over a 25-year period. Three groups of rectangles were defined, characterised by their annual rates of change in otter trawl effort: a group where otter trawl effort was declining; a group where effort was slowly increasing, between 0 and 199 hr yr^{-1}; and a group where effort was increasing rapidly, between 200 and 991 hr yr^{-1}. In both cases, differences between the groups were examined by one-way ANOVA (Figure 4.4.1.4.8). In each case, significant variation was detected, but in the direction opposite to that predicted by the hypothesis. Either fishing has caused an increase in species richness, or fishing has increased most in areas where species richness is highest.

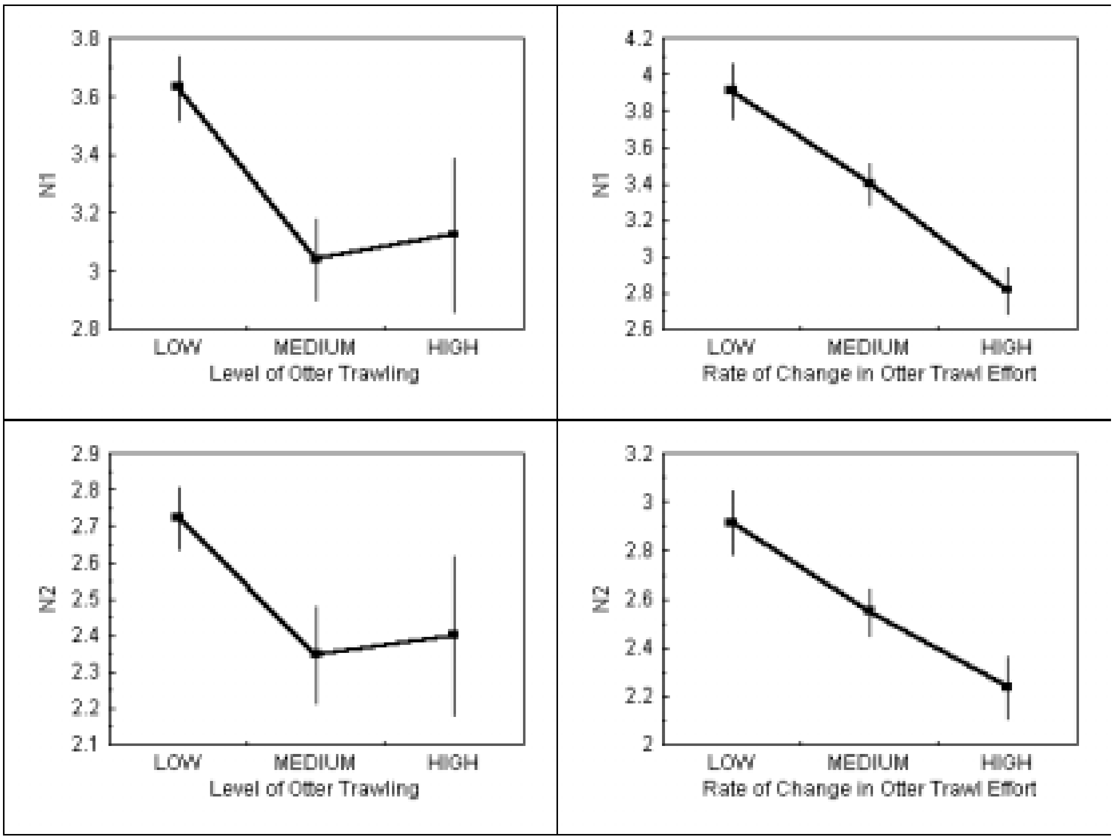
Figure 4.4.1.4.8. Variation in mean species richness (± 1S.E.) calculated for groups of rectangles varying in the level of otter trawling to which they were subjected between 1990 and 1995, and in which the annual rates of change in otter trawling differed over the period 1970 to 1994.



Hypothesis: Species diversity should be lower in areas where current levels of fishing effort are highest, and in areas where recent trends in fishing effort have shown the greatest increase.

The same treatments and analyses applied to examine the relationship between otter trawling and species richness were applied to two species diversity indices calculated from the pooled species abundance data for each of the 75 statistical rectangles. All ANOVAs were significant ($P<0.01$). In this instance, the responses of species diversity to variation in fishing effort lay in the anticipated direction. The difference in species diversity between areas of medium and high otter trawling intensity was marginal. However, both indices appeared to be sensitive to the full range of annual rates of change in fishing activity (see Figure 4.4.1.4.9).

Figure 4.4.1.4.9. Variation in mean species diversity N1 and N2 (± 1 S.E.) calculated for groups of rectangles varying in the level of otter trawling to which they were subjected between 1990 and 1995, and in which the annual rates of change in otter trawling differed over the period 1970 to 1994.



4.4.1.5 Summary of Scottish AGFS results and conclusions

Table 4.4.1.5.1 summarises the results obtained in the analyses carried out on the northwestern North Sea Scottish August Groundfish Survey data. These results clearly demonstrate that mean life history characteristics can detect effects of trawling. Again these results seem to suggest, as emphasised by WGECO in the past, that the application of a suite of metrics provides more information than any single metric alone. This is particularly true for the North Sea where fishing levels are high. Several of the life history characteristics appeared particularly sensitive to the effects of fishing at low fishing intensity. Under some circumstances, these metrics may not detect any further change in impact once otter trawl effort exceeds 5,000 hr yr^{-1}.

The results for the two Hill's (1973) diversity indices, N1 and N2, appeared to detect the effect of variation in fishing effort on species relative abundance in the groundfish community. They seemed particularly sensitive to changing levels of fishing disturbance. These data tend to corroborate the previous analyses of this data set. The response of species richness to variation in fishing impact was entirely contrary to the predictions of the hypothesis. Does fishing cause an increase in species richness? An alternative explanation is that fish abundance has increased most rapidly, and is now at high levels, in areas where species richness is greatest. This highlights the shortcoming of all analyses of this type—they are still not controlled experiments. However, more detailed analysis of this data set, taking account of other sources of variation, such as depth and other environmental factors, may still provide further insight into true cause and effect.

Table 4.4.1.5.1. Summary of results on Scottish AGFS data set (- no analysis, ✓ indicates a significant result, ✗ indicates result was non-significant, footnotes provide further commentary, LTb = slope of the long-term time series).

Independent variable or treatment	**Dependent variable or measure**	**Analysis**	**Parameter**						
			L_{inf}	**k**	**A_{mat}**	**L_{mat}**	**S**	**N1**	**N2**
Effort level	Parameter	Correlation	✗	✗	✓	✓	-	-	-
Rate of change effort	Parameter	Correlation	✓	✓	✗	✗	-	-	-
Effort level	Mean parameter	ANOVA	✓[1]	✓[1]	✓	✓[1]	✓[3]	✓[1]	✓[1]
Rate of change effort	Mean parameter	ANOVA	✓[2]	✓	✓[2]	✓[3]	✓[3]	✓	✓
Effort level	Parameter LTb	ANOVA	✓[1]	✓[1]	✓	✓[1]	-	-	-
Rate of change effort	Parameter LTb	ANOVA	✓[2]	✓	✓[3]	✓[3]	-	-	-

Cell entries are used to summarize patterns as:

1) Most differentiation was between low and medium levels of fishing effort. Data suggest either a threshold or a strongly non-linear effect. If used as a metric it may detect the effect of trawling impact as effort increases from low levels, but may not detect variation in impact as effort varies in relatively heavily fished areas.
2) Could be classed as insensitive—only differentiated between those rectangles where rates of change in effort varied from slow to fast increase. No difference between areas of slow increase and areas of decline in effort.
3) Significant ANOVA, but results difficult to interpret with respect to the hypothesis.

4.4.2 North Sea IBTS data

4.4.2.1 Species characteristics

A table of life history characteristics of fish species caught in the International Bottom Trawl Survey (IBTS) was modified from Daan (2001). Only a few of the life history characteristics could be obtained for the majority of the 266 species; these are maximum length, biogeographical area, habitat, lifestyle, and trophic level. For species with no maximum length (L_{max}) reported for the North Sea, the global value reported for that species was used. Habitat refers primarily to the water depth where the fish is found (e.g., shelf, slope) whereas lifestyle refers to where in the water column the fish is found (e.g., demersal, pelagic). Of these two, habitat was considered most useful for testing the above hypotheses. Trophic level was extracted from FishBase (www.fishbase.org) where it has been calculated from diet information or ECOPATH analyses of the ecosystems in which the given species live.

Additional life history characteristics that were recorded for a subset of the species include maximum age, age and length at maturity, L_{inf} and K from the von Bertalanffy equation, fecundity and egg size. However, these parametric estimates were only available for the subset of the species that are routinely sampled for age.

Ideally, each species could be ranked on an r/K continuum. One measure of the rate of increase (r) is the productivity parameter (∝) from a stock-recruitment relationship. Stock-recruitment relationships can be calculated for commercially important species for which SPAs are available, but not for the entire set of species found in the trawl surveys. Hall and Collie (unpublished) found an inverse relationship between the Ricker ∝ and L_{inf}. In this study L_{max} is used as an approximation for L_{inf}. In this manner, L_{max} is a surrogate for the rate of increase; species with low L_{max} are at the r end of the r/K continuum and vice versa, as is expected.

4.4.2.2 Survey data

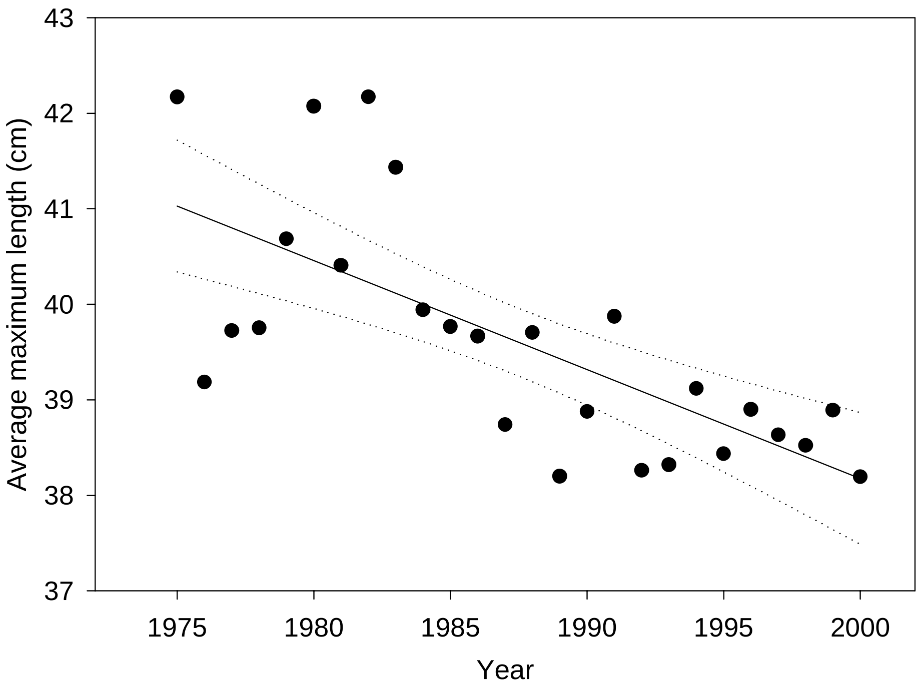
For the North Sea the hypotheses regarding the effects of fishing on traits of the fish community were tested using the International Bottom Trawl Survey (IBTS) data. The IBTS is a follow-up of the International Young Fish Surveys (IYFS) that were conducted in the North Sea and Skagerrak/Kattegat in 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 carried out by participating countries (ICES, 1999). 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. The GOV has a high vertical net opening of 5 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. For the present study, only quarter 1 data from the North Sea proper (excluding the Kattegat and Skagerrak) were used for the years 1974 until present. Each year only those hauls were used where all species caught were recorded.

4.4.2.3 Analysis and results

The hypothesis tested was that "species whose maximum length recorded in the entirety of any particular data set (L_{max}) should decline".

To assess the effect that life history strategy may have on the (changes in) abundance of populations, a life history index was developed based on a species' maximum length by weighting the biomass per species in the annual IBTS catch with the maximum length as expressed in Piet (2001). The average maximum length was shown to decrease significantly ($p<0.01$) from about 41 cm at the start of the sampling period to about 38 cm at the end of that period (Figure 4.4.2.3.1) indicating a relative increase of r-selected species.

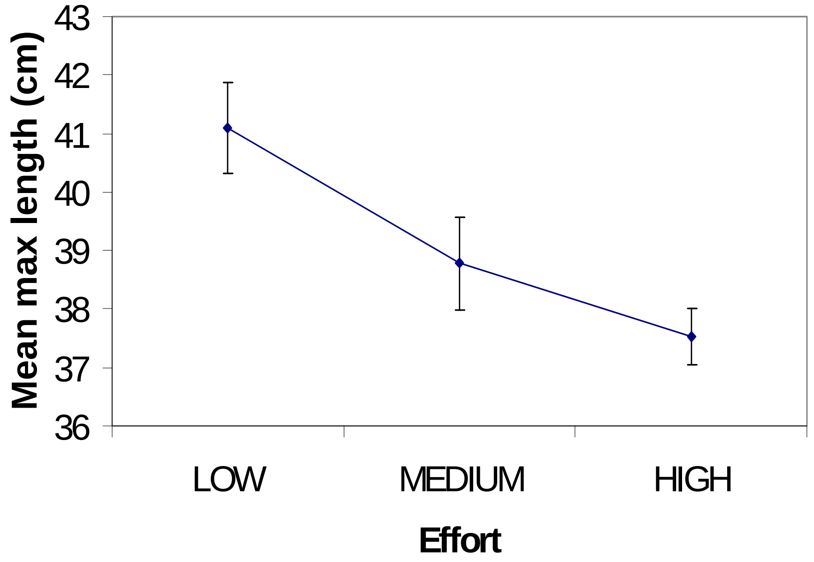
Figure 4.4.2.3.1. Average maximum length of the fish community over time. Points are values per year, lines show fit and 95 % confidence interval.



The effect of fishing effort on the maximum length index was studied by determining the mean maximum length and slope of the change in the maximum length index over time per ICES rectangle and combining these with the effort data per ICES rectangle according to Jennings *et al.* (1999a, 1999b, 2000).

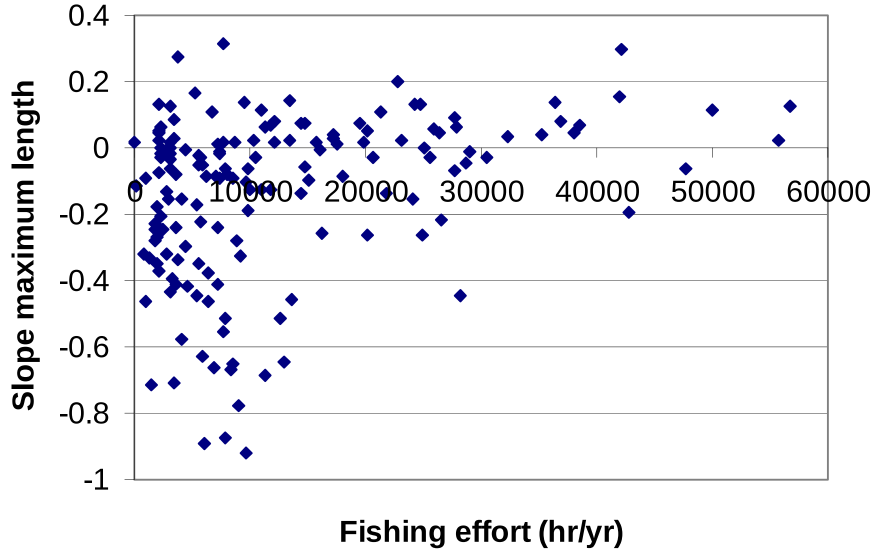
The mean maximum length showed a significant ($p<0.01$) decrease with fishing effort. Combination of ICES rectangles into three effort-classes (Low≤10,000, 10,000<Medium≤30,000, High>30,000 hr yr^{-1}) showed a significantly higher mean maximum length for ICES rectangles where fishing effort was "Low" (Figure 4.4.2.3.2).

Figure 4.4.2.3.2. Mean $Length_{max}$ and 95 % confidence limits for three classes of fishing effort (Low≤10,000, 10,000<Medium≤30,000, High>30,000 hr yr^{-1}). Based on 161 ICES rectangles for which IBTS and effort data were available.



The slope of the change in maximum length over time in relation to fishing effort is shown in Figure 4.4.2.3.3. Slope did not decrease with increasing effort as might be expected. The reason is that a significant inverse relationship between mean maximum length and slope was observed. This suggests that in the heavily fished ICES rectangles the composition of the fish community in terms of life history traits has stabilized at a relatively high level of r-strategists represented by a low mean maximum length.

Figure 4.4.2.3.3. Relationship between the slope of the mean maximum length over time and fishing effort in 161 ICES rectangles.



4.4.3 Portuguese survey data

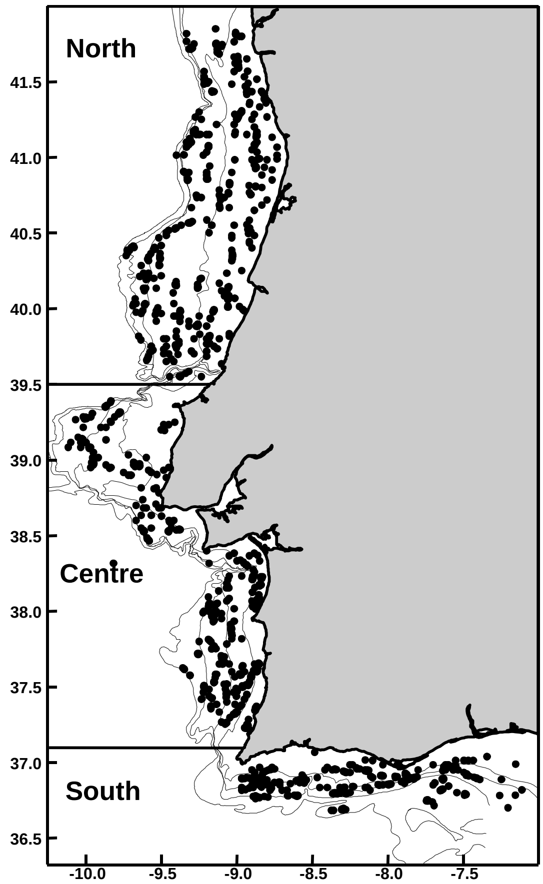
4.4.3.1 Species characteristics

Life history characteristics for 194 species caught in the Portuguese demersal survey were compiled by the Working Group. Of this group of species, 105 were common to the North Sea. An effort was made to standardize the sets of life history categories between regions such that these categories would be applicable to all regions of the North Atlantic. As for the North Sea, the variables that were available for most of the species were L_{max}, lifestyle, habitat and trophic level. These life history characteristics are global values for the species (extracted from FishBase, Whitehead *et al.,* 1984) and were not collected as part of the trawl survey. The 38 species for which one or more species characteristic were missing were excluded from the data set. These were all very rare species, collectively representing only 0.057 % of the total number of all individuals in the data set.

4.4.3.2 Survey data

Demersal survey cruises have been carried out annually in continental Portuguese waters since 1979 and are well described in Cardador *et al.* (1997) (Figure 4.4.3.2.1). Data from the autumn (fourth quarter) surveys for 1982 and from 1989 to 2000 are used here. All stations are separated into three geographic zones (North, Centre and South) at 39.5 ºN and 37.1 ºN latitude, and into two depth strata (less than and more than 150 m). This gave six groups of data that were used for all subsequent analyses. The criteria for these choices were largely taken from Gomes *et al.* (in press). For each year and for each of these groups, the total number of individuals of each species (after being scaled up to number of individuals for 1 hour for each haul, when necessary) was determined.

Figure 4.4.3.2.1. Map of continental Portuguese waters, showing survey stations (•), the three geographical zones (North, Centre and South) and with 100, 200, 500 and 750 m contour lines.



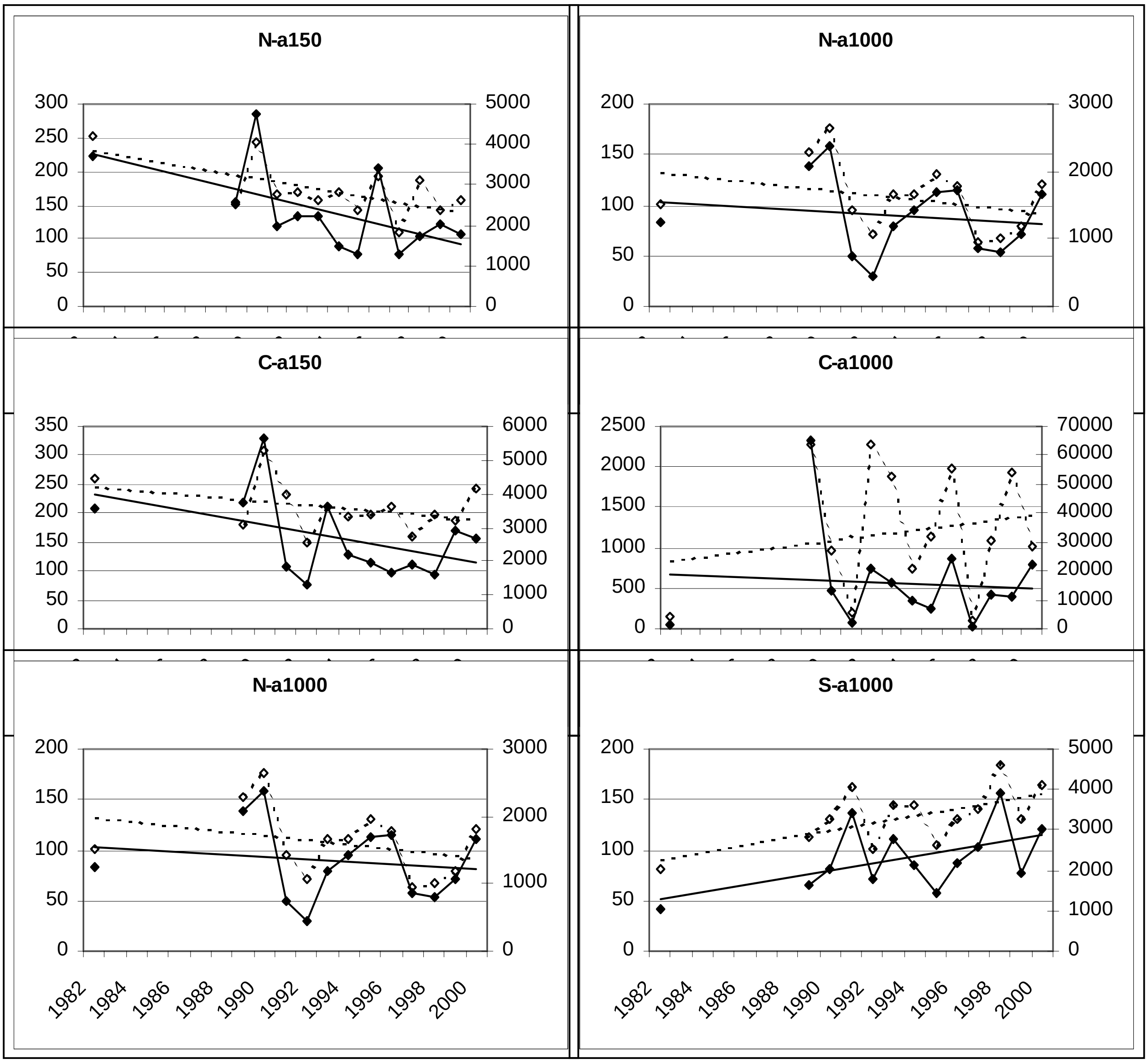
4.4.3.3 L_{max} and trophic level analysis and results

In order to test the hypotheses about L_{max} and trophic level presented above, the trends in these characteristics over time were analysed. This was done by calculating the weighted average value for each year based on the biomass of each species and the individual species characteristic value. These values were then plotted against time and trend lines determined (Table 4.4.3.3.1, Figure 4.4.3.3.1). From these data the following trends were observed, as shown in Figure 4.4.3.3.1.

Table 4.4.3.3.1. Trends (⇩ = decrease, ⇧ = increase) in weighted averages of L_{max} and trophic level. Values are R^2, a decrease with an R^2 over 0.25 is significant).

	L_{max}	Trophic level
North shallow	⇩ 0.335	⇩ 0.402
North deep	⇩ 0.022	⇩ 0.096
Centre shallow	⇩ 0.203	⇩ 0.130
Centre deep	⇩ 0.006	⇧ 0.036
South shallow	⇧ 0.137	⇧ 0.090
South deep	⇧ 0.278	⇧ 0.407

Figure 4.4.3.3.1. Plots of weighted averages of L_{max} and trophic level for 6 groups (N=North, C=Centre, S=South; a150=1–150 m, a1000=151 to 1000 m). Solid lines for L_{max}, dashed lines for trophic level.



It can therefore be seen that, except for in the north shallow group, these data do not show a significant decrease. There are a number of potential explanations for this, which could include:

- A decrease in exploitation since 1982;
- A change in faunal composition due to immigration of new species;
- The fishery targets small species;
- The shortness of the time series.

It is not surprising that there are differences between the north groups and the others as there are strong physical differences between these zones. The north has a relatively large and flat continental shelf. The northern and central zones are divided by the Nazaré canyon, and below this point the shelf is considerably more narrow.

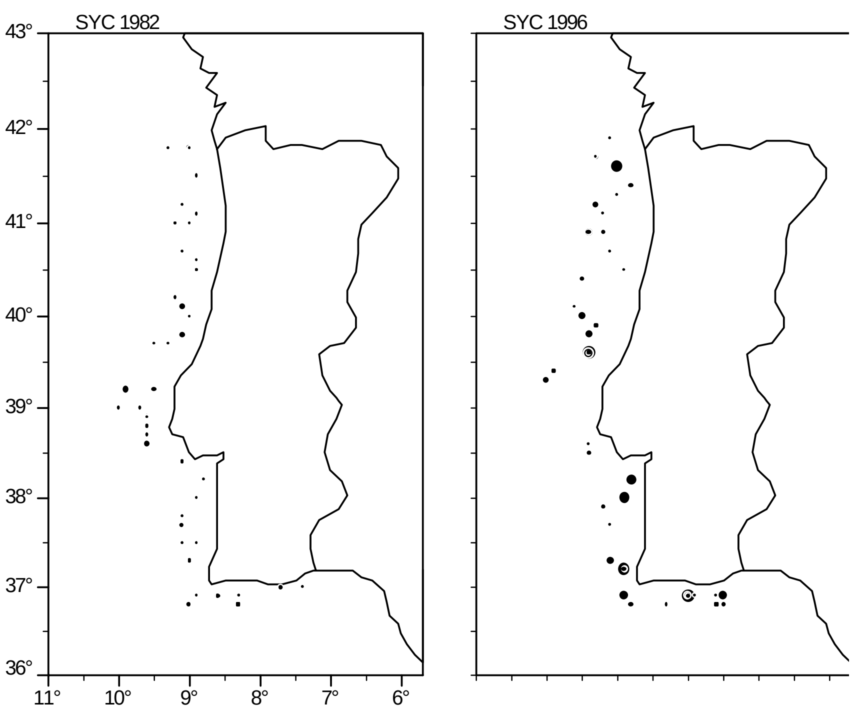
These analyses strongly suggest that these metrics are not reliable indicators on their own, at least not in this system. The life history traits are changing, but the patterns of change usually are not as predicted by theory, and the reasons for the changes are not understood. For example, in the centre deep group L_{max} decreased while trophic level increased. Both data sets contain a lot of noise, but trophic level more so, suggesting that trophic level is a less sensitive metric. Whether the differences between the results of these analyses and those using the Scottish data (Section 4.4.1), where a number of life history parameters were thought to have changed in ways consistent with theory, are due to differences between the ecosystems, between the fisheries, or just due to the differences in contrast within the data sets, remains to be explored.

4.4.3.4 Spatial metrics analysis

4.4.3.4.1 Description of data

The data were selected from the Portuguese survey data. Only data for elasmobranchs (27 species) and gadids (13 species from the Gadidae family) were used in order to keep the set small but to provide some contrast. Because of their reproductive strategies, elasmobranchs *a priori* may be expected to be more susceptible to effort. The distribution of the survey is shown in Figure 4.4.3.4.1. The subset had year, longitude-latitude, depth, species name, numbers caught, biomass caught, L_{max} and trophic level. The time series was for 1982 and 1989–2000. Preparatory to the analysis the data were aggregated to tenth of a degree squares and a code number given to each species to aid analysis. It would have been beneficial to apply the following analyses to other data sets, but time constraints prohibited this. In the following, we use the term community to denote either the elasmobranch or the gadid group.

Figure 4.4.3.4.1. Sample data from the Portuguese survey series. The figure contains the aggregated biomass for *Scyliorhinus canicula* 1982 and 1996. Larger circles reflect higher abundance.



Unfortunately, the data available (a subset of the Portuguese trawl survey) did not have sufficient quality (duration and knowledge of, or dynamic range of, effort) to test the effect of fishing effort on the metrics. Instead of a hypothesis-based study, a preliminary investigation of the performance of three spatial metrics on a single data set is reported.

4.4.3.4.2 Description of metrics

Because of the unavailability of effort data, the metrics were compared to abundance trends over the time series. Both the unweighted biomass per tow and L_{max} weighted biomass were considered.

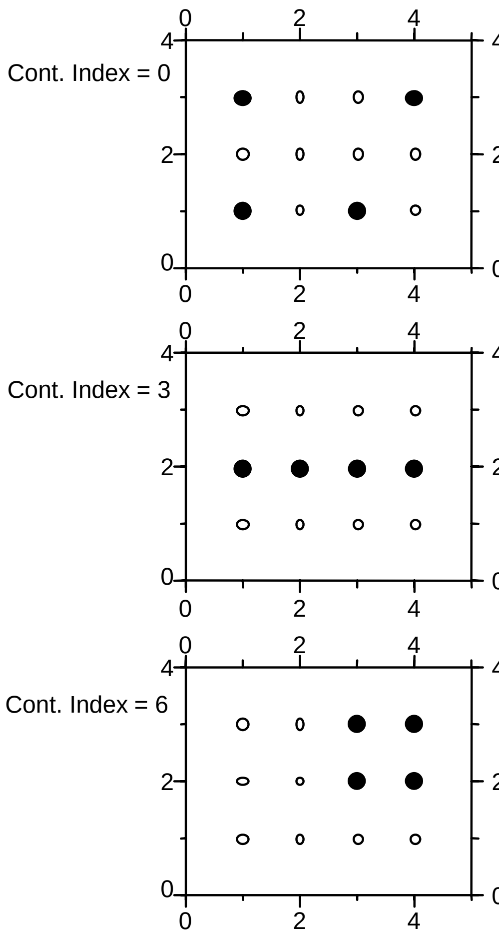
Spatial metric 1 – Anomaly of the center of mass of the community.

Based on experience from Eastern Scotian Shelf cod, a metric was proposed which is the anomaly of centre of mass of distribution. In that case it was noted that the centre of mass seemed to predict a subsequent rapid decrease in biomass in the late 1980s. The metric is calculated by first computing the centre of mass for each year and summing over the species or community under consideration. The average over time series of centroids is found and then the distance (in nautical miles) from each annual point to the average is found giving an annual anomaly.

Spatial metric 2 – Index of contagion.

A contagion index is proposed which is the number of neighbours within a set radius. Figure 4.4.3.4.2.1 shows four different distributions of four animals or sets of animals and a test radius of 2 units. In the upper plot, the four animals are so far apart that they have no neighbours within the test radius. In the middle plot, the four animals are in a row and there are three pairs of neighbours within the radius. Finally in the bottom plot, all four animals are in a cluster and the index is now 6. Because it is hypothesised above that contagion is probably more important on the species level, this metric is computed for each species of concern and then summed for all species under consideration. This sum could either be unweighted or weighted by abundance.

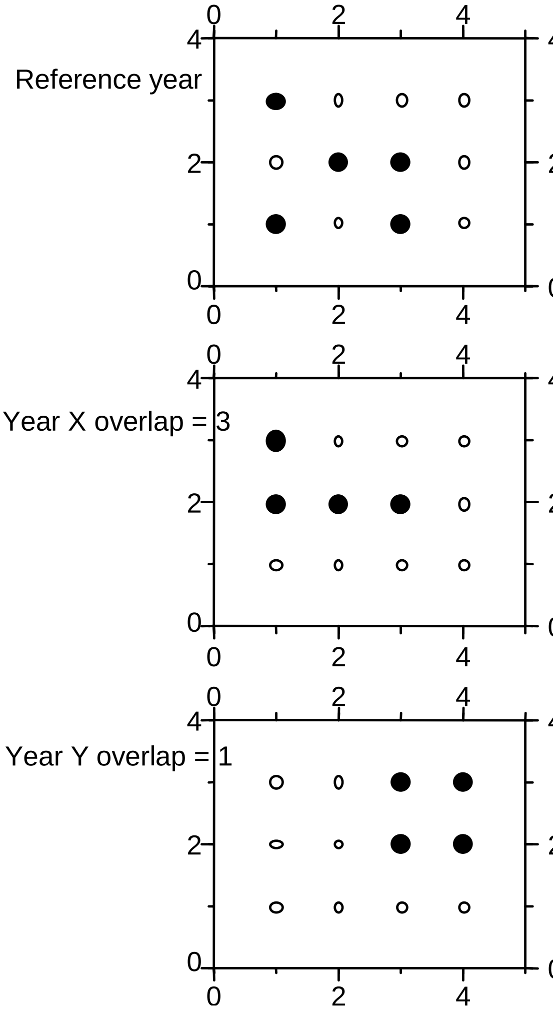
Figure 4.4.3.4.2.1. Samples of contagion index.



Spatial metric 3 – Index of overlap.

This index is proposed to indicate the displacement of a resource from its traditional, pristine or desired distribution. A reference year (or distribution) is chosen and then it is compared to the other years in a time series. As the data are aggregated onto a 0.1 degree grid it requires only to check if the same square is occupied as in the reference distribution. The index is the fraction of occupied grids in the reference distribution that are shared. Figure 4.4.3.4.2.2 shows this index for a simple data set.

Figure 4.4.3.4.2.2. Samples of an overlay index.



4.4.3.4.3 Analysis and results

For plotting and to ease comparison, all indices in the following have been normalised to their mean.

Figure 4.4.3.4.3.1 shows the abundance for the gadid group from the survey data as well as the abundance-weighted L_{max}. Abundance fell after 1982 while the weighted L_{max} showed little dynamics except for a small blip in 1992. The following figure (Figure 4.4.3.4.3.2) shows the abundance trends for animals with L_{max} above and below 100 cm. This shows that the larger animals were much more impacted over the data period but that the weighted L_{max} failed to pick up this event, probably because the larger group was such a small proportion of the total.

Figure 4.4.3.4.3.1. Gadid abundance L_{max} weighted by abundance.

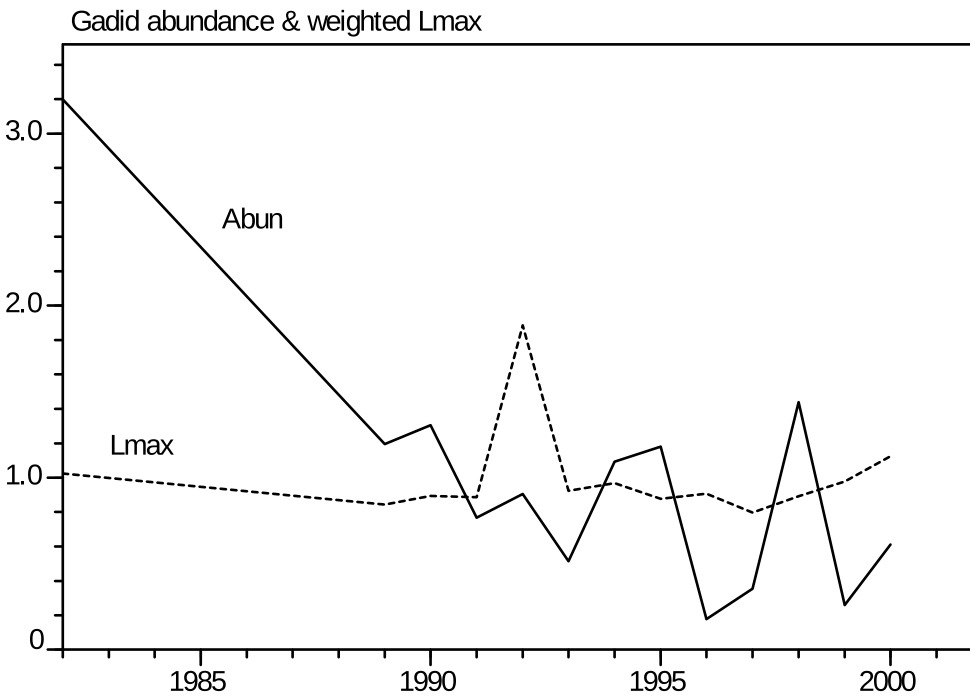
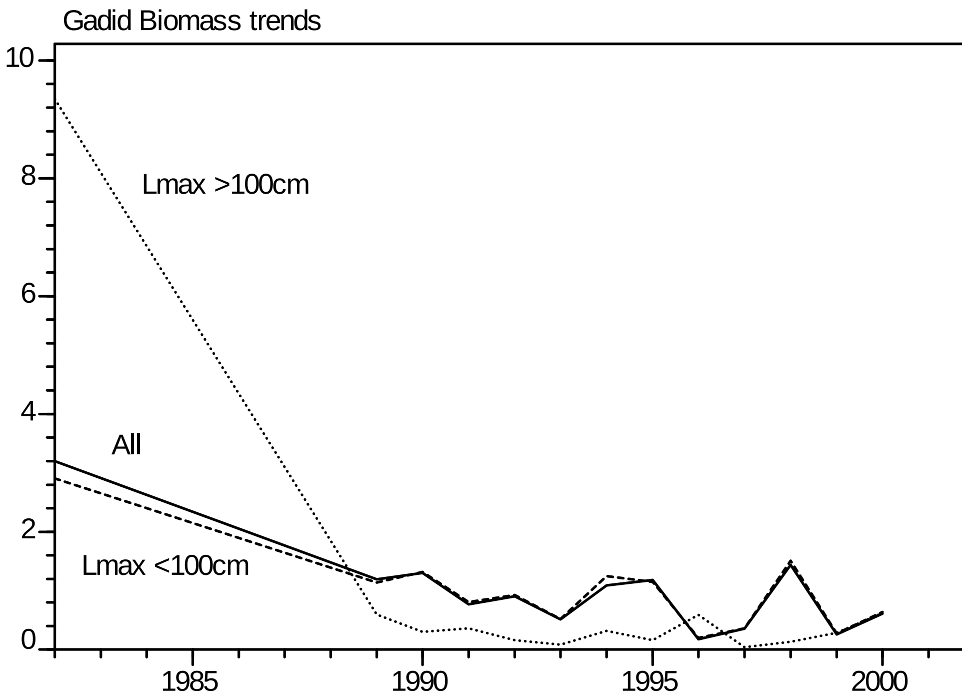


Figure 4.4.3.4.3.2 Gadid abundance after separating animals to those with L_{max} over and under 100 cm.



An analogous pair of plots are given for elasmobranchs (Figures 4.4.3.4.3.3 and 4.4.3.4.3.4). For this group of fish, there was a rise between 1982 and the next observation in 1989 in abundance which affected the weighted L_{max}. There was a spike in recruitment also in 1996 which did not affect the group's L_{max}, presumably because it was caused by animals near the mean L_{max}. The separation into size groups shows that the first event was due to large elasmobranches, while the second was dominated by smaller fish.

Figure 4.4.3.4.3.3. Elasmobranch abundance with and without L_{max} weighting.

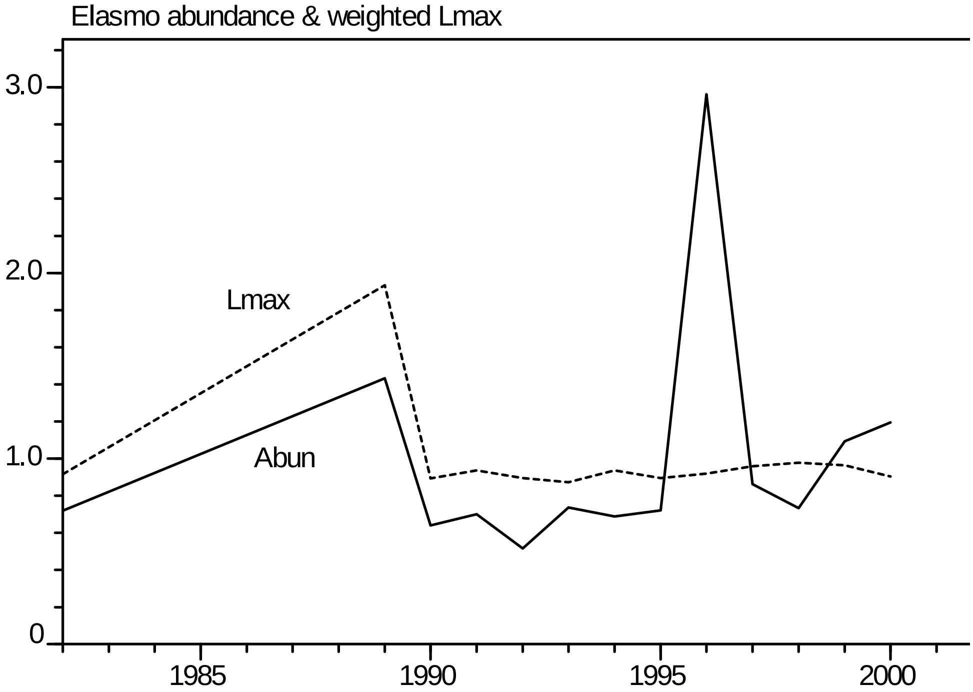


Figure 4.4.3.4.3.4. Gadid abundance after separating animals to those with L_{max} over and under 100 cm.

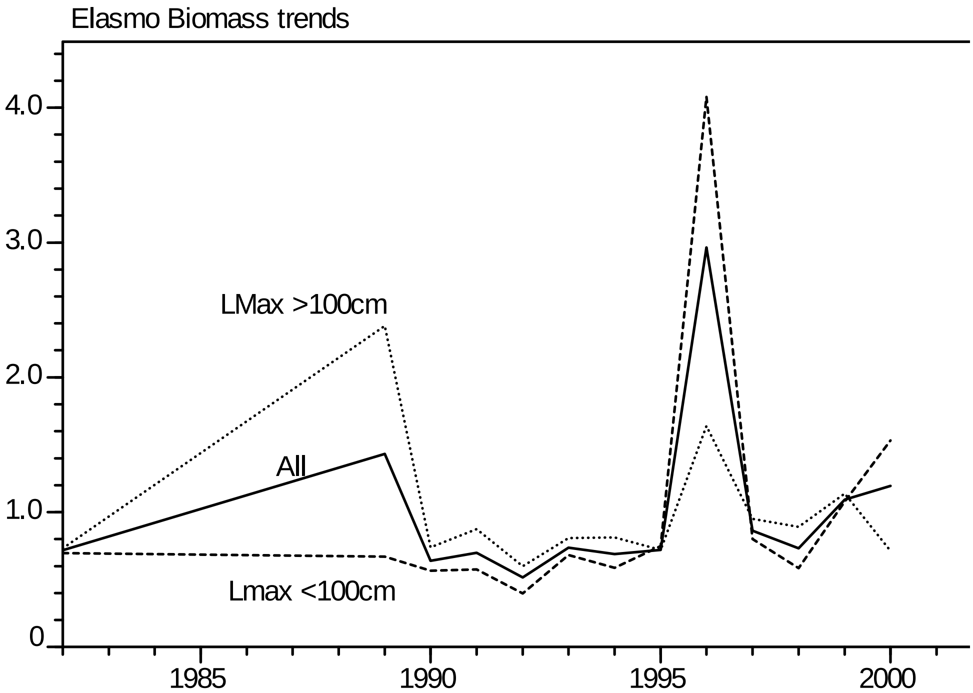


Figure 4.4.3.4.3.5 shows the abundance and the anomaly of the centre of biomass for the gadid group of fishes. It is difficult to infer the performance of the anomaly with this short data series although it appears to be somewhat opposite in phase to the abundance after 1989, suggested by the data from 1993, 1997, and 1998. Figure 4.4.3.4.3.6 shows a more dynamic anomaly. In 1992 there is little change in the biomass but the centre of mass moves dramatically, whereas in 1996 abundance shows a large change but the distribution metric changes very little. Further investigation is required to see if these reflect "real" events.

Figure 4.4.3.4.3.5. Gadid abundance and anomaly of the centre of mass.

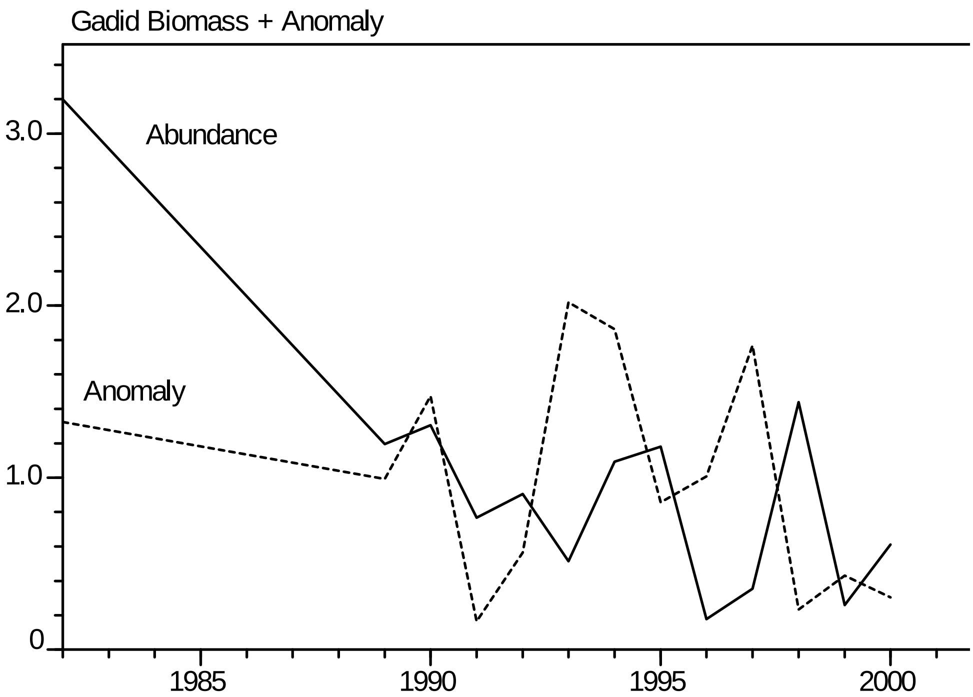


Figure 4.4.3.4.3.6. Elasmobranch abundance and anomaly of the centre of mass.

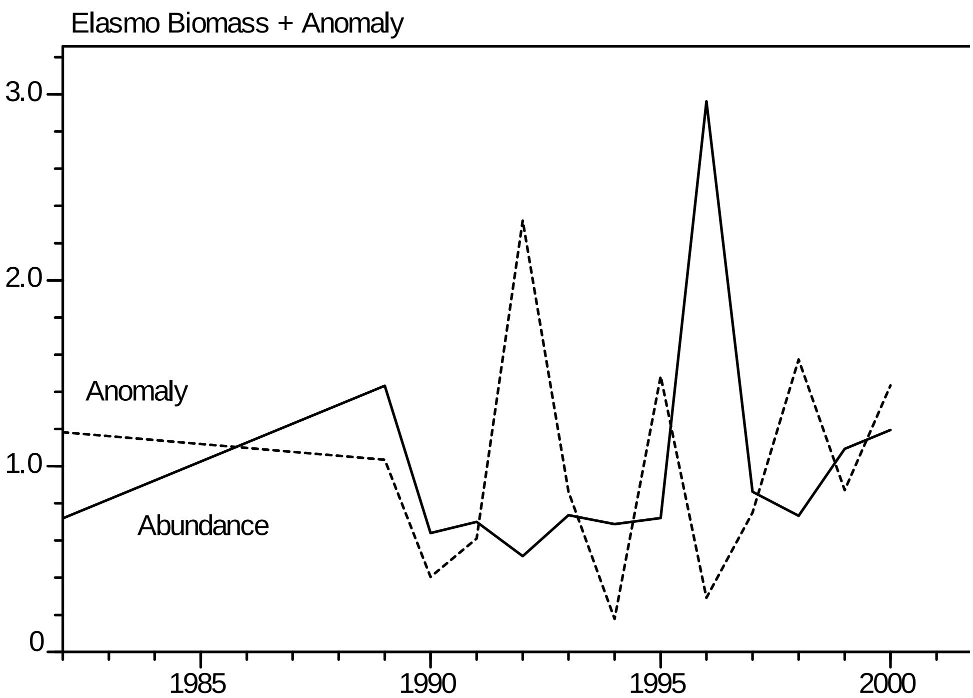


Figure 4.4.3.4.3.7 shows the abundance, contagion index, and overlap index for the gadid blue whiting (*Micromesistius poutassou*). The two spatial indices are highly correlated to the abundance. Figure 4.4.3.4.3.8 shows the abundance, contagion index, and overlap index for the elasmobranch *Scyliorhinus canicula*. These species were chosen because they were commonly seen in the survey. While the abundance has a spike in 1996, the spatial indices do not respond to the change, suggesting that the biomass distribution was not affected. Unlike the other two trends, the overlap fell after the reference year (1982), suggesting a displacement of the resource from that period.

Figure 4.4.3.4.3.7. Abundance, contagion index, and overlap index for blue whiting.

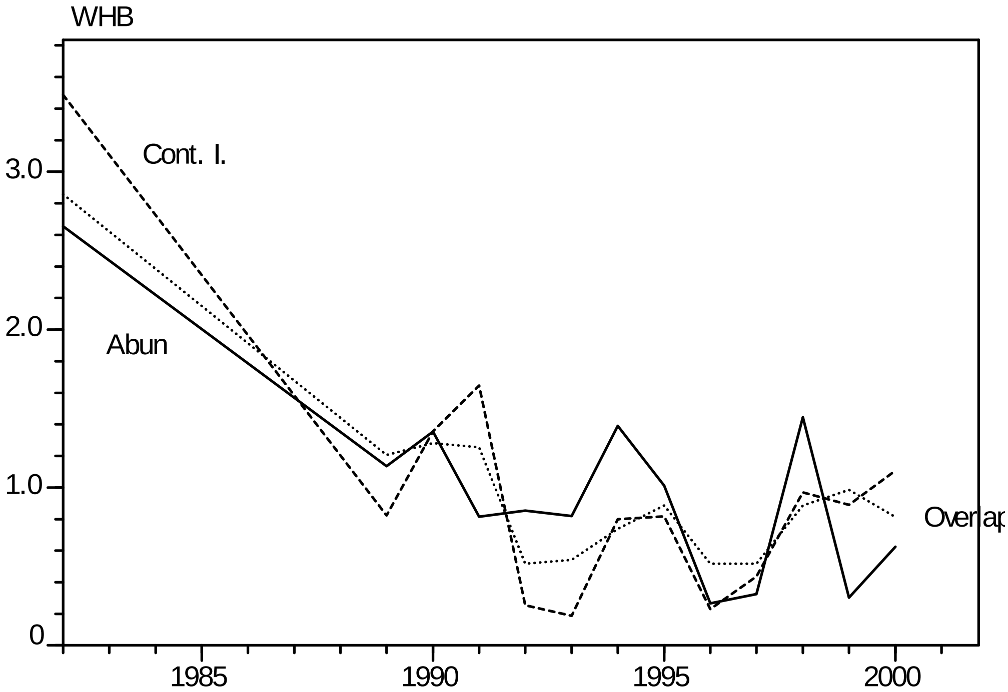
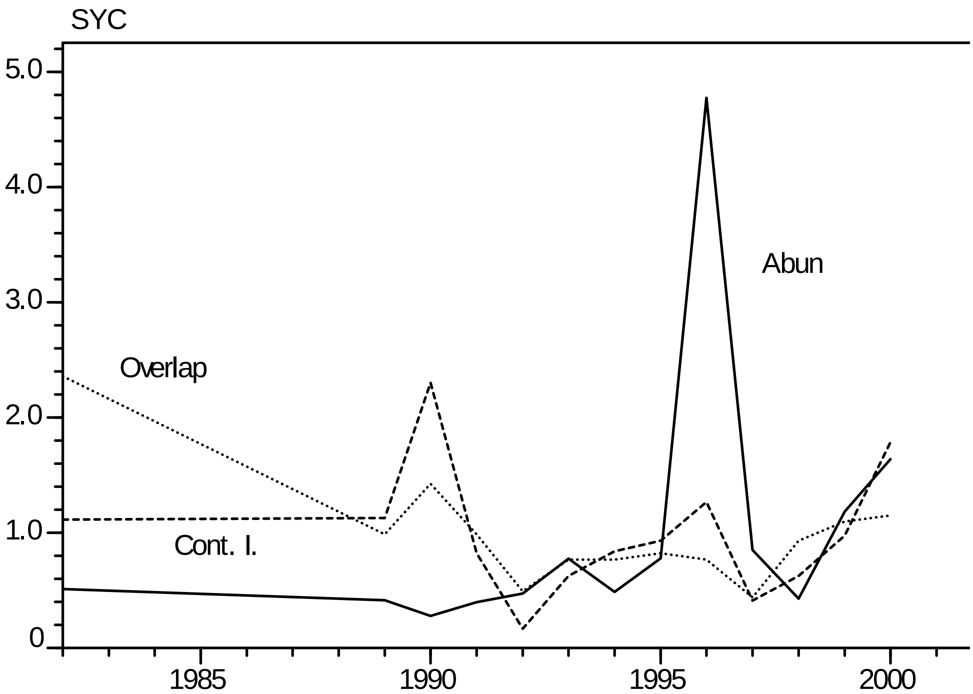


Figure 4.4.3.4.3.8. Abundance, contagion index, and overlap index for the elasmobranch species *Scyliorhinus canicula*.



The gadid group trends for abundance, contagion, and overlap are shown in Figure 4.4.3.4.3.9 and the three trends are quite similar. The indices for the elasmobranch group show more divergence (Figure 4.4.3.4.3.10). The overlap index fell even more than was seen in the single elasombranch species shown. The contagion increases with the increase in abundance in 1996, suggesting that for a number of elasmobranch species, the increase was localised.

Figure 4.4.3.4.3.9. Abundance, contagion index, and overlap index for the gadid group of species.

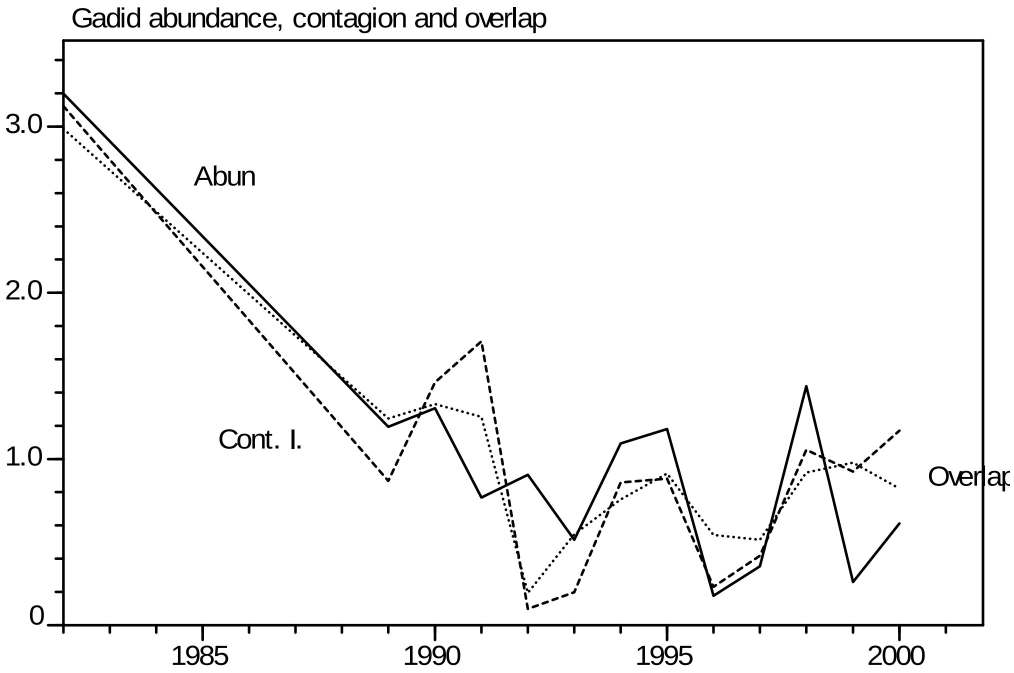
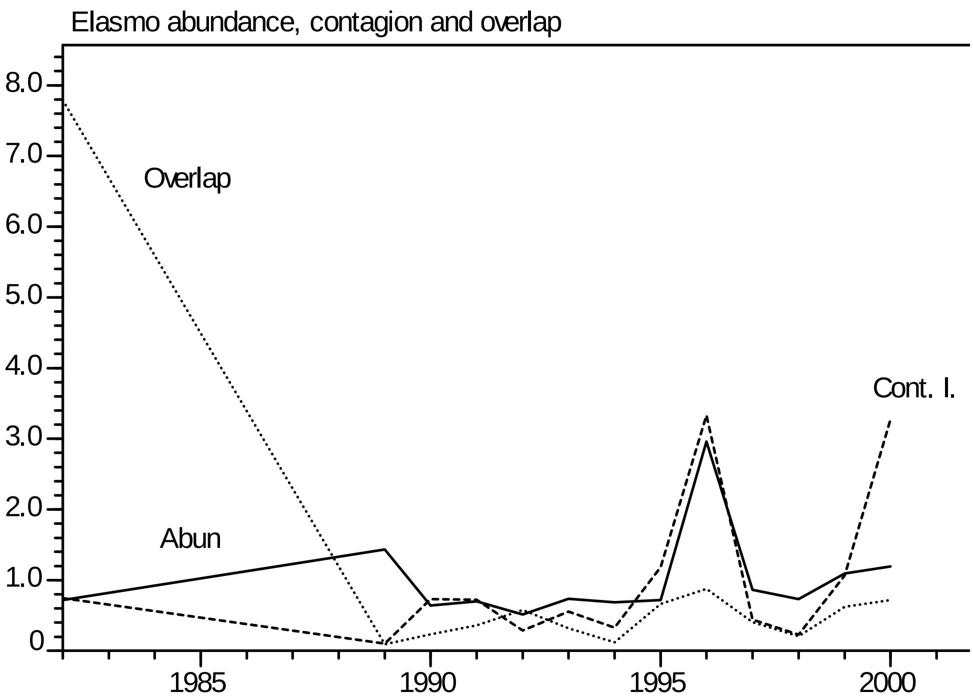


Figure 4.4.3.4.3.10. Abundance, contagion index, and overlap index for the elasmobranch group of species.



Thus, these results should be considered only as illustrations of the metrics. Even if all the analytical tools are working properly, the data are not sufficient to select among the proposed indices; none failed conspicuously, nor did any excel. Further research should be conducted in three areas: expansion to other sets of data; refinement of the metrics; and the development of a more methodical screening procedure.

4.5 Concluding thoughts and way forward

Many of these analyses were carried out in some haste and, while every effort was made to assure error-free analyses, time did not allow thorough review by WGECO. Furthermore, due to the same time constraints, not all the hypotheses proposed could be tested. Conclusions in these sections are preliminary, although it is important to highlight that many results are consistent with the predictions made about changes in life history characteristics. Nevertheless, at this stage the analyses were not detailed enough to be used to justify strong conclusions about the sensitivity and information content of life history traits relative to fishing effort. In particular, a better understanding is needed of how cases where predictions from life history theory were supported differed from the cases where either the predicted patterns were not found or where significant patterns actually were the opposite of the predictions from theory. WGECO feels that such analyses are important, because, along with impacts on physical habitat features, the effects of fishing on life history properties of species are some of the most lasting effects of fishing (ICES, 2000). WGECO would like to continue these investigations, but the most progress could be achieved if the meeting was not simultaneously working on Terms of Reference in support of ICES immediate advisory responsibilities. This issue is discussed further in Section 9 of this report.

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5 COMMUNITY-SCALE EqoQOs

5.1 An Introduction to Ecological Quality Objectives

5.1.1 History of EcoQOs

OSPAR and the North Sea Task Force (NSTF) have a relatively long history in the development of Ecological Quality Objectives (EcoQOs), as an approach to implementing the provisions of Annex V (on the protection and conservation of the ecosystems and biological diversity of the maritime area) of the 1992 Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR Convention) in the Northeast Atlantic (see Annex 2). Skjoldal (1999) gives a comprehensive overview of their evolution. Interestingly, the first call for a definition of terms of EcoQOs was in a draft of the European Commission Ecological Quality of Water Directive (Skjoldal, 1999). This is the ancestor of the EU Water Framework Directive that will become of growing importance for the management of coastal waters in the near future. However, the major starting point of EcoQOs has been the mutual demand of OSPAR and the North Sea Conferences for some method that allows assessment of the ecological status of the marine environment and defines objectives for the preferred ecological status. The basis for the concept was developed, beginning in 1992, during a sequence of three international workshops. Ecological Qualities (EcoQs) and the objectives derived from them have since been a permanent item on the OSPAR agenda, receiving regular attention during workshops and meetings. The result of all these efforts is that the scientific and political community connected to OSPAR began to develop and adapt a conceptual framework for EcoQs and EcoQOs. In some countries, additional scientific effort has been directed towards their further development.

In 1997, the basis was laid for the further advancement of the concept of EcoQOs through the Intermediate Ministerial Meeting on the Integration of Fisheries and Environmental Issues in the North Sea (IMM). During this meeting, both the Environmental and Fisheries Ministers composed a list of conclusions and recommendation on the integration issue. They are brought together in the Statement of Conclusions (IMM, 1997). Conclusion 2.6[1] calls for the development and implementation of an ecosystem approach in the management of marine ecosystems. As a follow up, a workshop on the ecosystem approach was held in 1998 in Oslo, Norway. This workshop concluded, amongst others, that clear objectives

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[1] The official text of Statement of Conclusion 2.6:

2.6 further integration of fisheries and environmental protection, conservation and management measures, drawing upon the development and application of an ecosystem approach which, as far as the best available scientific understanding and information permit, is based on in particular:

- the identification of processes in, and influences on, the ecosystems which are critical for maintaining their characteristic structure and functioning, productivity and biological diversity;
- taking into account the interaction among the different components in the food-webs of the ecosystems (multi-species approach) and other important ecosystem interactions; and
- providing for a chemical, physical and biological environment in these ecosystems consistent with a high level of protection of those critical ecosystem processes;

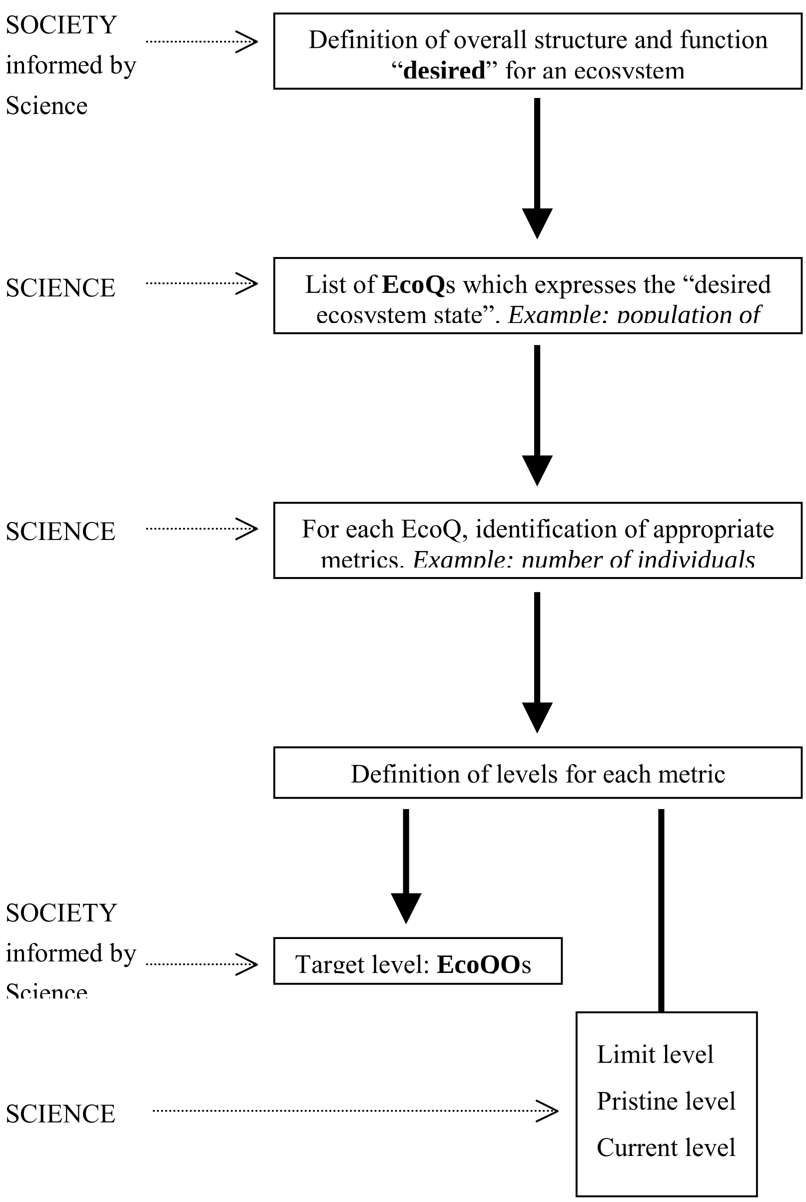
are needed as part of the development of an ecosystem approach. The workshop further suggested that Ecological Quality Objectives under development within OSPAR could provide a solid basis for defining clear objectives (Anon., 1998). As a result a workshop specifically on Ecological Quality Objectives was organised in 1999 in Scheveningen, The Netherlands. Both workshops were attended by a mixture of policymakers, stakeholders, and scientists.

The basic ecosystem properties included in the OSPAR conceptual framework for EcoQs (Skjoldal, 1999) are:

- Diversity;
- Stability;
- Resilience;
- Productivity;
- Trophic Structure.

Because EcoQs have to address ecosystem properties in relation to human influences, the OSPAR JAMP (Joint Assessment and Monitoring Programme) issues were taken as a basis for covering the latter. Together, these issues make up the conceptual framework shown in Figure 5.1.1.1. Habitat issues were a late addition.

Figure. 5.1.1.1. Conceptual framework for the methodology of describing EcoQ and setting EcoQOs (from Lanters *et al.*, 1999).



Based on a document especially prepared for the meeting (Lanters *et al*., 1999), the stakeholders, policymakers, and scientists present at the Scheveningen workshop concluded that EcoQOs should be developed for ten issues (Anon., 1999). These ten issues cover EcoQOs at the species, community, and ecosystem levels. They also more or less cover the range from structural (diversity) to functional (processes) aspects of the ecosystem. OSPAR agreed that this list of ten issues would form the basis for future work (OSPAR, 2000a), but would also keep an open eye for further improvement or extension of the proposed list of issues.

Table 5.1.1.1. The proposed set of ten issues for EcoQOs for the North Sea derived from the Scheveningen workshop (Anon., 1999).

Proposed set of issues	
1	Reference points for commercial fish species
2	Threatened or declining species
3	Sea mammals
4	Seabirds
5	Fish communities
6	Benthic communities
7	Plankton communities
8	Habitats
9	Nutrient budgets and production
10	Oxygen consumption

The set of ten issues is currently being further explored under the guidance of OSPAR. Norway, the Netherlands, ICES and the OSPAR Eutrophication Task Group (ETG) are taking care of the ten issues according to the following plan:

1) Commercial fish species	Norway
2) Threatened and declining species	Netherlands
3) Sea mammals	ICES
4) Birds	ICES
5) Fish communities	Netherlands
6) Benthic communities	Netherlands/ETG
7) Plankton communities	Norway/ETG
8) Habitats	Norway
9) Nutrient budgets and production	ETG
10) Oxygen	ETG

The main objective of the OSPAR Biodiversity Committee is to see whether it is feasible to put some very clear examples of EcoQOs on the agenda of the Fifth North Sea Conference in March 2002. In this process, ICES is responsible only for the elaboration of EcoQOs for marine mammals and seabird species. All other issues fall outside the official request of OSPAR for ICES advice. Nevertheless, this does not mean that EcoQOs are of no importance for the ICES community. First of all, ICES could provide OSPAR with a first independent evaluation of the scientific credibility of the framework and methods being applied. Furthermore, ICES has a long history of dealing with ecological reference points that will be of great value when newer fields of marine science are explored.

In 1997, on a related issue WGECO was asked to: "*Develop and examine potential reference points which might be used for including ecosystem considerations in relation to the precautionary approach*". This request was approached by considering whether the reference points already developed for commercial fish species offered sufficient conditions to ensure effective conservation of the larger ecosystem, if management were to respect the reference points fully. This approach was justified with the reasoning that, although a few conceptual and many operational problems remained with advising on and managing fisheries in a precautionary framework, the tasks were still much simpler, and the practical experience greater, with marine fisheries management than with marine ecosystem management (ICES, 1998). WGECO concluded that, to ensure conservation of the ecosystem, additional reference points were required for:

- non-target species (by-catch and gear damage effects);
- ecologically dependent species (predators dependent on harvested species);

- species affected by scavengers (whose abundance increased by feeding on discards and offal);
- genetic diversity of exploited species.

When the list was completed, it was observed that conservation of each of these ecosystem components could be achieved through additional single-species (or nutrient-specific) reference points, where the species were carefully chosen on ecological grounds.

Reference points beyond species level were considered in depth, but intentionally not brought forward for two reasons. First, community- and ecosystem-scale reference points were thought to be too speculative, because there was insufficient practical knowledge and theoretical basis for identifying limit or precautionary reference points. Second, notwithstanding the diverse modelling expertise in the Working Group, no member was able to propose an integrative property of the ecosystem that could be shown to be at risk, if the component species were being individually conserved with high probability. Both of these reasons highlighted the need for further study, because ecosystem reference points are potentially interesting and it was suggested that the use of models may help in understanding the behaviour of ecosystem metrics (ICES, 1998).

The issue of ecosystem objectives was revisited during the 1999 meeting of WGECO. The necessary objectives for ecosystem conservation were made more specific, to include spatial properties of populations as well as their abundances or biomasses. More attention was also given to objectives for the conservation of habitat features. However, with regard to emergent properties of ecosystems, WGECO again concluded that "While not ruling out the need to continue to monitor developments in this area, WGECO finds no evidence that such ecosystem properties need, or even can be, subject to direct management objectives. However, WGECO acknowledges that, even if reference points for emergent properties are not warranted by present knowledge, many measures of ecosystem properties, such as measures of diversity, can serve a valuable role in communicating with many clients of marine science" (ICES, 2000a).

5.1.2 Terminological issues

Both OSPAR and ICES have been trying to place scientific advice and management decision-making with regard to marine environments and resources into a more rigorous and explicit framework. These efforts, and those of many other groups worldwide, have evolved from the meetings and agreements following from the 1992 UN Conference on Environment and Development (UNCED) in Rio, so it should not be surprising that many terms and phrases are used by both OSPAR and ICES (and other marine conservation and management organizations). Unfortunately, the terms have been evolving partially independently (even within different parts of ICES), so similar words and phrases often mean different things when used by different bodies. This creates potential for confusion and misunderstandings. The involvement of ICES with OSPAR's initiative to develop EcoQOs for the North Sea makes it particularly important that terms be used in consistent and clear manners (ICES, 2000c, 2001d; OSPAR, 2000a). Although there has been a small evolution in the definition of EcoQs and EcoQOs, the main features of their definitions have hardly altered since 1992. Because EcoQOs are currently being developed under the flag of OSPAR, the definitions that came as a result of the Scheveningen Workshop (Anon., 1999) will be used. The following definitions apply throughout this report (ICES usages are those used throughout all ICES advice on fisheries, as summarised in Section 1 of ICES, 2001d):

Ecological Quality (EcoQ): An overall expression of the structure and function of the marine ecosystem taking into account the biological community and natural physiographic, geographic and climatic factors as well as physical and chemical conditions including those resulting from human activities.

Ecological Quality Objectives (EcoQO): The desired level of ecological quality relative to a reference level.

Reference points: In **ICES** advice regarding fisheries, **reference points** are specific values of measurable properties of systems (biological, social, or economic) used as benchmarks for management and scientific advice. They function in management systems as guides to decisions or actions that will either maintain the probability of violating a reference point below a pre-identified risk tolerance, or keep the probability of achieving a reference point above a pre-identified risk tolerance (ICES, 2001d). There will be multiple reference points for any single property of a system, each serving a specific purpose. In advice on non-fisheries issues, ICES terminology has been somewhat more variable, with **reference value** sometimes used in contexts identical to those where reference point is used in advice on fisheries. The ICES Management Committee on the Advisory Process (MCAP) should resolve these terminological inconsistencies as quickly as possible.

Reference level: In **OSPAR** usage, **reference level** began as the level of EcoQ where the anthropogenic influence on the ecological system is minimal. It became clear that it could be very difficult or impossible to determine such reference levels, when systematic monitoring of properties related to the EcoQ began well after pristine conditions were perturbed. This not only applies to biological conditions, but also to naturally occurring chemical substances. Therefore, OSPAR acknowledged that a pragmatic approach may be required to establish and use reference levels. OSPAR noted that temporal trends could be informative about past conditions, and in some circumstances preliminary reference levels could be taken as the starting point of a time series. For this reason, the wording "**a** reference level" was preferred over the use of "**the** reference level" in the EcoQO definition (Anon., 1999). It should be emphasised that "reference level" should not be confused with the objective. Although the original meaning of "reference level" as defined in the context of EcoQOs had a different meaning than "reference points" used in the context of fisheries (OSPAR, 2000a), the modified usage by OSPAR leads to the meaning of **reference level** being specific to each application. OSPAR and ICES seem to still differ somewhat because, at least for the present, within OSPAR there appears to be only a single reference level per EcoQO at any time. It appears that the criteria on which the reference level is set can change from EQ to EQ, or over time, leading to changes in the reference level as well, so in that sense reference level does function much like the concept of reference points in ICES advice.

Target Reference Points: In **ICES** usage, particularly for fisheries, 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. Where they address biological aspects of ecosystems, target reference points must in all cases be at least as "safe" as precautionary reference points selected on exclusively biological considerations. Beyond that conservation-based constraint, ICES has stressed that managers, decision-makers, and stakeholders have the responsibility for selecting target reference points (see Section 5.1.3). When ICES provides advice relative to **target reference points**, unless otherwise requested ICES assumes that management should be designed to achieve them on average, and hence advice is risk neutral with regard to them, as long as conservation reference points are not placed at unacceptable risk.

Targets levels: In **OSPAR** usage, target levels identify states of the EcoQO (or, operationally, values of the metrics of the EcoQO) that management should be trying to maintain with high probability. In this usage, they function in a manner very similar to Target Reference Points as used by ICES. However, the request from OSPAR to ICES, as a scientific advisory body, to provide advice on suitable target levels suggests that target levels are identified through scientific endeavours. This is quite different from the ICES perspective on target reference points (see Section 5.1.3), and the difference has not yet been resolved.

Limit Reference Point: In **ICES** usage, 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. Outside the limit reference point, the stock has entered a state where there is evidence that:

- productivity is seriously compromised, <u>or</u>
- exploitation is not sustainable <u>or</u>
- stock dynamics are unknown.

Management should maintain stocks inside limit reference points with high probability. To account for uncertainty in assessments, ICES uses **precautionary reference points** as a basis for scientific advice, with the intent that management consistent with precautionary reference points should have at least a 95 % probability of keeping a property away from its limit reference point. Limit Reference Points are based on the biology of the stock/species/ecosystem, independent of social and economic considerations. Hence ICES has argued that they should be identified by technical experts, and has selected limit reference points for stocks on which it provides scientific advice.

OSPAR does not appear to have chosen to include the notion of limit reference points within the EcoQ and EcoQO framework that it is developing.

The request of OSPAR to ICES to develop EcoQOs makes it clear that the sometimes subtle differences in philosophies behind these concepts and terms needs to be understood clearly. In the following text the philosophy behind the use of reference points within ICES fisheries advice is explained (taken from ICES, 1997):

"Reference points are a key concept in implementing a precautionary approach. The following points from Annex II of the UN Agreement on Straddling Fish Stocks and Highly Migratory Fish Stocks are relevant to the distinction between target and limit reference points:

2. Two types of precautionary reference points should be used: conservation, or limit, reference points and management, or target, reference points. Limit reference points set boundaries which are intended to constrain *harvesting within safe biological limits within which the stocks can produce maximum sustainable yield. Target reference points are intended to meet management objectives.*

3. Precautionary reference points should be stock-specific to account, inter alia, for the reproductive capacity, the resilience of each stock and the characteristics of fisheries exploiting the stock, as well as other sources of mortality and major sources of uncertainty.

5. Fishery management strategies shall ensure that the risk of exceeding limit reference points is very low. If a stock falls below a limit reference point or is at risk of falling below such a reference point, conservation and management action should be initiated to facilitate stock recovery. Fishery management strategies shall ensure that target reference points are not exceeded on average.

7. The fishing mortality rate which generates maximum sustainable yield should be regarded as a minimum standard for limit reference points. For stocks which are not overfished, fishery management strategies shall ensure that fishing mortality does not exceed that which corresponds to maximum sustainable yield, and that the biomass does not fall below a predefined threshold. For overfished stocks, the biomass which would produce maximum sustainable yield can serve as a rebuilding target."

Therefore, reference points stated in terms of fishing mortality rates or biomass, or in other units, should be regarded as signposts giving information of the status of the stock in relation to predefined limits that should be avoided or targets that should be aimed at in order to achieve the management objective.

Although not points of specific inconsistency between OSPAR and ICES, there are a few terms used in very specific and consistent ways in ICES fisheries advice, but in the larger community of those interested in marine ecosystems and conservation the terms have a variety of meanings. In this report the terms will always be used with the ICES meanings, unless specifically stated otherwise. For that reason it may be helpful to explain those usages here:

Conservation is used in the sense of conserving natural resources. The resources can be used as long as the usage is at rates and in ways that do not place the resource, or the ecosystem in which it is found, at risk of harm that is serious or difficult to reverse in the short, medium or long term. Resources may be being conserved when they are in conditions quite different from their pristine states.

Sustainability is used to refer to the use(s) made of the resource, and not to the state of the resource. A strategy for use of a resource is sustainable when it could be pursued in the long term without causing unacceptable risk of a conservation problem for the resource being used, or the ecosystem in which it is found. Quite often a fishery, for example, is said to be sustainable, when, to be precise, what is meant is that the strategies used to manage and prosecute the fisheries are sustainable. By applying "sustainable" strictly to the use, and not to the resource itself, this is a slightly more restrictive use of the term "sustainable" than is encountered in some general reports on conservation of biodiversity, but is in no way inconsistent with those uses.

For example, the Convention on Biological Diversity (CBD) defines the term Sustainable Use to mean "the use of components of biological diversity in a way and at a rate that does not lead to the long-term decline of biological diversity, thereby maintaining its potential to meet the needs and aspirations of present and future generations." As with the ICES usage, the CBD definition includes the notions of using the resource, but in ways that can be continued in the long term without causing conservation problems.

The final terminological issue relative to this report is our use of **metric** to refer to the biological attribute that is being considered as an indicator of an ecological quality of the system. In our discussions, we routinely used "indicator" and "metric" interchangeably. However, in the written report, WGECO took note that "indicator" sometimes carries a specific meaning as an "indicator species". Therefore, we decided to use **metric** in all cases where we mean something that can be measured quantitatively (or, when appropriate, qualitatively) and is at least be considered as being a suitable way to measure the ecological property that the EcoQ is intended to capture. Where we use **indicator,** we mean for it to be interpreted in the sense of "indicator species".

5.1.3 Conceptual issues

5.1.3.1 Interaction between EcoQ and EcoQO

The requirement for the development of EcoQOs arises from the need to bring forward an "ecosystem approach" to environmental management. This is a key part of the adoption of the Convention on Biological Diversity (CBD) signed at the UN Rio Conference and adopted as a basis for management by the EU and the Intermediate Ministerial Meeting of North Sea ministers.

Unfortunately, the term "ecosystem approach" has been used in a wide variety of contexts and been imparted with a range of definitions, as have the terms EcoQ and EcoQO (Section 5.1.2). From the OSPAR definitions, a sequential framework for developing EcoQs and EcoQOs can be seen (Figure 5.1.1.1). The starting point for the development of ecosystem approaches to environmental management is to define the "overall structure and function" desired for the ecosystem being considered. The specification of this "desired ecosystem" is a societal decision, although science has some key roles (see Section 5.1.3.2). This desired overall state of the ecosystem must be expressed as a series of clear statements that will constitute the list of EcoQs. Next, it is necessary to identify at least one metric for each EcoQ. The question of the necessary and sufficient number of metrics to ensure conservation of the system, or even achieve the EcoQs specified by society, is not simple (Section 5.1.4). From this list of metrics, one must derive desired levels for various measures of the system, which correspond back to the "desired ecosystem" initially specified by society. The desired values of the metrics comprise the suite of EcoQOs. Consistent with the changing OSPAR definition of "reference level", there is no inherent need for EcoQOs to be set always to the condition where anthropogenic influences are minimal. In fact, this would imply no use of environmental services such as waste treatment or food production. Rather, the "appropriate" values for the EcoQOs are determined by the overall desired ecosystem. The appropriate measures and quantitative values for the EcoQs and EcoQOs will vary among systems and depend on the priority given to various issues. Moreover, it is implicit that the setting of EcoQOs should be done in an integrated manner, to ensure that they are mutually achievable and collectively sufficient to ensure conservation of the ecosystem. However, for pragmatic reasons the initial approach used at the Scheveningen workshop and continued by OSPAR in its request for advice is to develop EcoQOs for various ecosystem components in a variety of different groups (Section 5.1.1). The implications of a number of these issues will be discussed in Section 5.3.

5.1.3.2 Role of science

The different approaches to reference points, reference levels, limits, and targets increases the potential for confusion about suitable roles for technical experts, policymakers, and advocates of many sectors including users and non-users. Although it is inappropriate for ICES to advise on preferred governance approaches among policymakers and public sectors, it is important that the role of science be understood in the larger process of selecting and implementing EcoQs and EcoQOs. Note that the term "technical expert" is used here to make clear that "scientists" includes not just biological, physical, and chemical scientists and collaborating quantitative experts. Social sciences also have an important contribution to make to the role of science.

The selection of properties of ecosystems that are essential to their conservation is the responsibility of technical experts, as is the selection of metrics of those properties. If clients wish to have relative priorities assigned to the general properties or their specific metrics, technical experts also have a key, but not exclusive, role. Technical experts are the appropriate group to assign priorities based on the degree to which conservation of the ecosystem depends on each of various properties of the system, as well as to assign priorities among metrics based on their reliability and sensitivity. Rankings of properties and metrics based on human values is not an issue appropriate for biological and physical scientists, although social scientists may work with policymakers and the public to clarify public opinion on such rankings.

Once a suite of properties needed for conservation of the ecosystem is identified, and metrics of the properties have been selected, several groups have roles in setting various benchmarks along the metrics, and identifying acceptable and unacceptable domains of the properties. It is the responsibility of the technical experts to specify lower (or upper) conservation limits for metrics and properties; that is, values of a metric or states of a property below (or above) which there is increasing risk of harm that is serious or difficult to reverse. (Some properties and their metrics may have both upper and lower limits associated with conservation.) There will almost always be uncertainties with regard to determination of both conservation limits of properties and metrics, and current states of properties and metrics. Technical experts are also responsible for quantifying such uncertainties to the fullest extent possible, and selecting precautionary positions on the properties and metrics such that if management is risk neutral relative to the precautionary reference points, there will be a high probability that the conservation limits will be avoided. For many plausible candidate metrics, there is insufficient contrast in the historical data (if the data exist at all) to be informative about where the conservation limit may be, and in such instances, technical experts have special challenges to determining how to advise on managing risk.

If policymakers or the public wish to know the state of a property prior to substantial anthropogenic perturbations, it is also a question that should be answered by technical experts. That does not mean that the question always is answerable, or that the answer, if possible to provide, is a sound basis for management. The same points apply to questions about the maximum value (or minimum) that a property or metric could assume, if management were intended to achieve the most extreme state possible for that ecological attribute of a system.

Between the states that are determined by conservation limits to be avoided with high probability and the most unaltered or extreme value possible to achieve, policy makers and society have to choose the desired state that management should aim for. Such targets are chosen on the basis of society's values, often as interpreted by policy-makers. Technical experts may participate in this exercise as citizens, advocating whatever point of view they may have. However, they have the responsibility to acknowledge that they are merely advocating their particular special interest (even if they believe it is an especially enlightened one), and have no special privileges at the table where competing interests are seeking consensus. It can be difficult to keep these identities distinct, because the technical experts have a role during the negotiations leading to setting management targets: that of warning when targets under consideration would place the conservation limits at unacceptable risk of being violated. Such advice has to be perceived as objective and impartial, which can be hard when the same individuals have been involved in debates over proper values to be the basis for society's choices. Assuming that consensus can be achieved on a set of management objectives that are mutually compatible, the technical experts have a final role to lead the translation of society's values, often expressed qualitatively, into operational management targets, expressed in the currencies of the metrics. This may make it appear that the technical experts are setting the targets, or the EcoQOs, but their role is only as translator of society's choices onto the biological axes that are being used.

5.1.3.3 Approaches to setting EcoQOs

5.1.3.3.1 Approaches used by other Working Groups or experts

WGECO had available draft text on EcoQOs from the Working Group on Seabird Ecology (WGSE) and the Working Group on Marine Mammal Population Dynamics and Habitats (WGMMPH) (Section 6), and OSPAR consultants' reports on EcoQOs for benthos and threatened and declining species. WGECO began by examining the approaches taken for these four ecosystem components and considered them with a view to developing a generic context for determining EcoQOs.

In the cases we examined (all biological systems), it was recognised that it was impossible to know what the pristine state of a system which has minimal anthropogenic influence should be. For contaminants, it is relatively easy to see what the reference level (*sensu* OSPAR prior to 2000) should be, i.e., zero for synthetic substances such as DDT, PCBs, and the appropriate biogeochemically determined level for naturally occurring substances. This is not the case for biological populations or communities.

The benthic reference level proposed (de Boer *et al*., 2001) is that it should "represent the situation under minimal human impact". The report then advocates the use of values derived from the 1986 data series as a basis for EcoQOs (although it is noted that these should be regarded as minimum/maximum values for the proposed metrics), thereby implying that the situation in 1986 is the acceptable ecological quality. The WGSE and WGMMPH were concerned with EcoQOs for these species groups and the EcoQOs proposed reflect this emphasis. The WGSE considered two possible approaches, the possibility of defining metrics for each species which give a measure of ecosystem health, i.e., using each species as an ecosystem metric, or the development of metrics of possible impacts which use appropriate aspects of seabird ecology. WGSE proposed the latter as being a more sensible approach and so developed EcoQOs relevant to eight ecosystem anthropogenic effects that use seabirds as metrics.

WGMMPH generally concurred with the approach of WGSE, expressing concern, however, that the WGSE approach did not give sufficient prominence to population size, which they considered to be the trait of most relevance to the public. They developed a hierarchical figure, illustrating a series of steps from population size, through life history factors such as productivity and mortality, to a list of human effects from the OSPAR JAMP, and discussed the relationships that could possibly exist among the effects, the life history factors, and ultimately population size. They also discussed the concepts of target and reference levels on EcoQ metrics. In the documentation available at the end of the formal meeting of WGMMPH, specific EcoQs and their metrics had not been identified, however. Rather it was reported that they would continue to pursue the ideas behind the tabulation. It was expected that most or all of the EcoQs and their metrics would be derived from important life history and biological properties of marine mammal populations, and subsequently linkages would be sought to the human effects. This is somewhat in contrast to the approach of WGSE, who began with the ten issues identified by OSPAR, and then sought properties of seabirds considered particularly sensitive to each.

For the "threatened and declining species" the objective is more clear—the rebuilding of populations, although the level to which they should be rebuilt, i.e., 50 % of the reference level, the target EcoQO needs to be determined within a societal framework. The key issue here was what criteria triggered inclusion of a species as "threatened and declining";

an issue that although in concept is exclusively scientific, in practice is hotly debated among even scientific experts (see Section 7.4).

5.1.3.3.2 Major influences on WGECO's approach

In Section 5.3, the approach WGECO followed in selecting possible EcoQs and their metrics is explained in detail, and its application is illustrated in Section 7. As much as possible, WGECO adhered to the spirit of the EcoQ initiative as it was understood. However, there are a couple of important considerations which arose in discussion.

First, for reasons explained in Section 5.1.3.2, WGECO is not proposing any EcoQOs for any EcoQs. This group, or other groups of scientists, could provide estimates of ecologically defined positions on the metric of an EcoQ, and inform on the ecological consequences of positions along the EcoQ metric that society may be contemplating using as an objective. However, science groups have no basis for actually choosing the position that society desires on the metric.

Second, this Working Group, and ICES in general, has established its scientific credibility through applying rigorous scientific standards for its advice. The scientific concepts and tools of integrated ecosystem management may start off as somewhat more abstract and much more complex than those used in management focused on a single target species in a fishery or a single contaminant. Likewise, the data and models available for use in setting and monitoring status against EcoQs and EcoQOs may be even more incomplete, contain more sources of uncertainty, and be, at present, less well tested. WGECO did not use these realities as an excuse to lower scientific standards for advancing the EcoQ initiative. This does not mean that a good scientific basis cannot eventually be available for supporting integrated ecosystem management, whether or not EcoQs and EcoQOs are the tools that are used. For now, however, it is important to make the reliability of the scientific basis for progress as clear as possible to those outside the community of experts. When scientific advice is requested on a specific issue, including on a specific ecosystem quality however poorly studied, WGECO, and ICES in general, will provide the best advice possible, pointing out uncertainties and potential weaknesses. In this case, however, WGECO interpreted its task as being asked to make what progress was possible on identifying community-scale EcoQs and suitable metrics for them, maintaining the usual scientific standards of WGECO and ICES.

5.1.4 Issues regarding implementation

5.1.4.1 Lessons learned from past experience

The ongoing development of EcoQOs for the North Sea in various fora, as well as the specific OSPAR requests to ICES to provide recommendations for "appropriate" EcoQ indices for marine mammals and seabirds, evoked a discussion on the added value of this approach, from a scientific standpoint, compared to existing management objectives.

Several existing policies to regulate the effect of anthropogenic impacts on the marine environment have been successful, for instance in diminishing nutrient loads and various sources of pollution. However, at present fisheries are broadly, and probably rightly, seen as having by far the most important impact, not only on commercial fish stocks but also on the ecosystem at large (OSPAR, 2000b). Most target species of North Sea fisheries are overfished, even though in practice, the nature of the overfishing problem is well known. Fisheries science has developed over many years to provide a rigorously defendable advisory framework, wherein the advice provided meets high standards for objectivity, peer review, and consistency (Section 5.1.3.3). The advice is primarily based on evaluating the necessary and sufficient conditions for conservation and sustainable exploitation of commercial stocks, using carefully screened data sets and assessment models. Studies of the advice have found patterns of systematic overestimation of future biomass and underestimation of exploitation rates in many fish stocks (van Beek and Pastoors, 1999), indicating that the models and/or data were not perfect. However, even where quantitative details of the scientific advice on fish stock management have been inaccurate or imprecise, technical experts have consistently advised management actions that would have moved the fisheries in the direction of greater sustainability (Serchuk *et al*., 2000). Nonetheless, overharvesting has continued and for many species the situation has become worse since the Common Fisheries Policy (CFP) was adopted in 1983.

After so long a period with limited progress on eliminating overfishing, it is important to consider what factors have contributed to the lack of progress on a clearly identified and scientifically tractable objective (reduce overfishing). Limitations on fisheries science, the current management system itself, and the current decision-making environment for fisheries are thought to have contributed to the ongoing problems. Limitations on fisheries science may have contributed to continued overfishing directly through the inaccuracies referred to above and indirectly through creating openings for opponents to argue for deferment of action pending greater certainty. The TAC-based management system as presently applied may be intrinsically unsuitable to control fishing mortality on an annual basis. The failure of the system may be partly attributed to TACs having been set too high (EC, 2001), partly to ineffective enforcement and intentional failure of harvesters to comply with management plans, and partly to the multispecies nature of fisheries, which cannot hit several TAC targets simultaneously. In decision-making about fisheries, opponents of fishery restrictions are well organized at least at the local level, know the political system well, and have exploited uncertainties and even small errors in assessments to discredit advice and delay implementation. Given the institutional problems in the policy setting and management of fisheries, even perfect assessments would not guarantee an effective TAC management regime (Daan, 1997).

In the Green Paper on fisheries, the EC now suggests that the solution for failing TAC management may lie in making multiannual and multispecies TACs (EC, 2001). In such a management system, however, any scientific predictions of such quantities will require even more complex models and analyses. These will have an even higher degree of uncertainty than the annual species-specific catch options currently calculated by assessment working groups, and greater opportunity for errors that may not be detected before the advice is provided. Thus, while not making major improvements to other management system and decision-making factors that contribute to overfishing, the scientific advisory challenges have been made greater.

These developments have two important implications. First, to the extent that weaknesses in past scientific advice contributed to the failure of the CFP to achieve sustainable harvesting, future scientific advice has the potential to contain even more such weaknesses. Some steps to use lessons from the past to shore up these potential weaknesses are discussed in section 5.1.4.2. Second, to the extent that the management system and decision-making process for fisheries are at fault, they require major overhauls. It is in this pessimistic context that the application of the EcoQ and EcoQO initiative to fisheries problems must be viewed. In promoting the ecosystem approach, the Inter-Ministerial Meeting on the North Sea and OSPAR have initiated development of an integrated policy for the conservation of the marine environment. This policy will be debated and enacted in a public opinion climate strongly influenced by public and political frustration over the ineffectiveness of the Common Fisheries Policy to control fishing pressure (Green Paper), as well as concern over the future consequences for the marine ecosystem, should the present situation be allowed to continue indefinitely.

The integration of all relevant management policies, including fisheries, within a single framework is an intrinsic component of an ecosystem approach to management. Such integration makes setting fisheries policy part of a much larger debate, where the legitimacy of many more stakeholders and concerns is indisputable. Placing debates on fisheries policy in this larger framework may mobilize social and political support for conservation issues, and alter the management and decision-making climate that has failed to prevent overfishing in the last few decades. Even without structural changes the greater support may strengthen the will of policy makers to make effective decisions to reduce overfishing, and the ability of managers to implement and enforce those decisions. However, without structural changes to the management systems that address directly the reasons why the existing legal framework failed to restrict overfishing the benefits of adopting a much broader approach of defining a coherent set of EcoQs and EcoQOs may not be achievable. It is of great concern that EcoQs and EcoQOs are not mentioned in the fisheries Green Paper, which suggests that OSPAR and EU may be on different tracks with their policy development. The different tracks invite questions about the degree of commitment of fisheries managers to move their policy development and management into this larger and more socially inclusive framework of ecosystem management. The institutional changes needed to ensure this transition occcurs are also discussed in Section 5.1.4.2.

In summary, unless fisheries management is brought within the framework that OSPAR is developing, it will not be possible for OSPAR to achieve the goals which motivated it to pursue the EcoQ framework. However, even if fisheries were to come within the framework, many of the reasons why overfishing has continue would not be addressed.

5.1.4.2 Applications of lessons from history to the Advisory and Management System needed to implement EcoQ-based management

As noted in Section 5.1.4.1, the management system within the marine environment has failed in a number of areas. The greatest area of failure that has had an effect at a basin-wide scale has been in fisheries management (OSPAR, 2000b). If ecosystem-based management is to be implemented, consideration of the effects of all human activities on the ecosystem needs to be integrated at the highest policy level. At present, the management of fisheries in the North Sea (and in the wider EU area) is carried out by fisheries ministers who are responsible both for conservation of fish stocks and for promotion of the fishing industry. Policymakers and managers for fisheries are responsible for setting (and accountable for achieving) both conservation objectives for fish stocks and socio-economic objectives for fisheries. Adequate structures or mechanisms are not in place to reconcile discrepancies that arise now between either the conservation and socio-economic objectives within fisheries, or in future between conservation objectives for fish stocks and the more encompassing integrated ecosystem objectives. The decoupling of those responsible for setting and delivering conservation objectives from those responsible for setting and delivering socio-economic objectives is one possible step towards a system where more integrated ecosystem management could be pursued. This would still not resolve the problems presented by the absence of mechanisms to reconcile discrepancies among objectives set for fisheries conservation and those set for integrated ecosystem management, were any to occur (Symes and Pope, 2000). In fact, it might reveal a need for a mechanism to reconcile discrepancies between objectives set for conservation of fish stocks and socio-economic objectives set for fisheries. If the current fisheries policy and management framework in the North Sea were merely provided with objectives relating to the ecosystem derived by OSPAR, institutional changes to increase the accountability of managers to meet those additional objectives might be needed as well, in order to have a high likelihood of achieving more integrated ecosystem-based management and better management of fisheries.

Applying the past experience of WGECO, a number of needs and opportunities for improvement of the science and advisory systems can be identified. If ICES is to be involved in 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. As noted in Section 5.1.4.1, despite strict protocols, great collective experience, and high vigilance, occasionally poor data and some errors in stock assessments escape the review by both working groups and advisory committees. Although it is possible at this stage to define and propose metrics that meet the available selection criteria and, combined, may provide a broad picture of the health of the system (Section 5.5), any metric may be calculated from a variety of available data sets that have not been collected for this particular purpose. Moreover, subtle variations in algorithms for calculating indices may sometimes have a significant influence on their performance. Give that EcoQs and EcoQOs, once adopted, are altered only periodically, recommending a particular metric is technically demanding and more complex than it may initially appear. Once a metric and reference levels on it have been selected, review and advisory groups with the skills of the best assessment working groups, but even greater breadth of knowledge and expertise, will be essential if management based on the EcoQOs is to have a sound scientific foundation.

There are clearly far more potential metrics of EcoQs that could be used in management of the North Sea than are practical, given available funds for monitoring and assessments. OSPAR will have to make some choices among them, but once made, there are a number of science activities that must be done. Scientists should carry out a sensitivity analysis of various methods and data sets to select on technical grounds the optimal combination for future use. This step alone may require further interaction with OSPAR, if the detailed technical review reveals unforeseen but crippling technical problems for some preferred metrics of ecological quality. Once EcoQ metrics, data standards and calculation algorithms all have been decided upon, relevant data sets for each of them must be collected and analysed periodically. Both processes require quality control to ensure that any advice derived from such data is perfectly defendable.

There is still considerable uncertainty about the effectiveness with which such metrics may in practice measure the response of the system to human impact. Therefore the research community should work with the science advisory and management framework explicitly to explore the occurrence of true hits as well as false alarms and misses in historic series of the EcoQ metric and human activity. Also, it is important to ascertain that the metrics match the set of potential impacts that management measures can address, and to evaluate the performance of EcoQO-based advice over time in improving management decision-making and actions.

Once the metrics have been selected, monitoring and analyses completed, the results subjected to peer review, and advice developed, the scientific advice will be given to a management system which has thus far proven unable to solve the relatively simpler problem of controlling overfishing, given advice on the fishery and target stocks. Even with the structural changes discussed above, there are specific problems of science advice that should be addressed:

1) The selection of "appropriate" EcoQOs is not straightforward (Section 5.1.3), partly because what is "appropriate" cannot be singularly defined scientifically, and partly because there is incomplete scientific knowledge about what aspects of an ecosystem are necessary and sufficient for its conservation. Compared to single-species fisheries advice, where keeping spawning biomass large, and exploitation rates low, is likely (but not guaranteed) to keep harvesting sustainable and to conserve stocks, guides to successful ecosystem management are less clear. Given the complexity of marine ecosystems, there are many properties that one might argue need to be conserved and a nearly infinite number of potential metrics of these properties. It is clear from a pragmatic point of view that we have to be selective, and have to select wisely. Although it is relatively easy to formulate important selection criteria for EcoQ metrics (Section 5.4), applying these over a wide scale of potential metrics is by no means straightforward.
2) More importantly, the approach chosen by OSPAR deviates from the existing one for commercial stocks, because in the OSPAR framework the EcoQO (the target) is to be set relative to the current level and to a reference level that should reflect a situation when anthropogenic impact was minimal (with allowance for a pragmatic approach), rather than a limit reference point (LRP) referring to conditions considered not sustainable and posing unacceptable risk to the resource (Section 5.1.2). In fact, for many potential EcoQ metrics it will be hard, if at all possible, to define a level associated with "unsustainability" or otherwise with an unacceptable threat to the ecosystem. In the EcoQ system, the possibility of large numbers of metrics combined with poorly determined conservation limits on many of them will make any scientific advice even easier to contest by stakeholders and also by other experts. Current

fisheries advice formulated in the sense of keeping the impact below some unsustainable level is obviously much easier to defend than EcoQ-based advice that points to some current and historic values whose distances from a LRP are known only vaguely or not at all. The resultant lack of defensibility might well further reduce rather than enforce the impact of scientific advice on management and therefore could easily undermine the advisory role of ICES.

3) By definition, any broad EcoQ metric for a community reflects the ecosystem response to a broad set of human impacts, and therefore the contribution of each activity to its present value may not be singled out easily. In fact, any particular value of a metric of an EcoQ may arise from completely different combinations of different impacts. This will make it much more difficult to predict how the metric will respond to various options to reduce one particular impact, and to assign responsibility (and associated costs) among possible contributors, when a metric does indicate a conservation problem. On these grounds, EcoQs and their metrics selected because they are responsive to a specific threat seem particularly useful (although see Section 5.1.3.3.1).

Although the approach seems promising in principle, embarking on giving advice on EcoQOs will set high demands on developing a rigorous and defendable advisory framework, which will take considerable time. Therefore, it would seem wise to concentrate on developing a suite of EcoQ metrics first and to test their performance particularly with a view to defining potential LRPs before endeavouring recommendations on EcoQOs. It is likely that management systems, as well as science advisory systems, must also adjust to new and greater demands on their effectiveness, if they are to be able to enact and enforce management measures based on the best ecosystem advice possible.

We cannot know now the detailed organisation and procedures for the management system that will actually create and implement the management policies and plans based on the scientific advice regarding status of ecological features relative to their target levels, as measured by the metrics and EcoQOs. However, that process must function much more effectively than the current one, for progress to be made on the pieces (the individual EcoQs) and for this process to actually result in effective ecosystem management, leading to improved ecosystem quality.

5.1.4.3 Practical considerations regarding making EcoQs work together for integrated management

The OSPAR decision to proceed with identifying EcoQs separately for ten issues permits possibly hundreds of EcoQs to be proposed, in order to guarantee that the entire marine ecosystem and all the processes that operate within it were covered. Although this decision was considered to be pragmatic (Scheveningen Workshop, Anon., 1999), each EcoQ would have at least one EcoQO to be monitored and managed. Currently, fisheries managers struggle to address adequately targets for 14 annually assessed commercial fish and benthic species in the North Sea, along with the additional seven non-assessed species, or species groups, for which TACs are set. Add to these the need to account simultaneously for EcoQOs for threatened and declining species, seabird and marine mammal species, fish and benthos communities, habitats, and two ecosystem process issues, and the task of managers becomes much more complex. Where management actions will be necessary, some may be difficult, costly, and/or controversial, and for reasons of logistics or politics, it may not be possible to implement them all at once. This creates at least two classes of problems: assigning priorities and achieving intercompatibility.

The requirement to rank these EcoQs and EcoQOs so as to be able to choose which to pursue aggressively and which to defer, seems inevitable. Where much effort has been invested in gaining social consensus on EcoQOs on which different sectors of society placed different initial values, and the achievement of which will demand differential subsequent costs, opening a second debate on the priority of that EcoQO relative to others may be divisive. It needs to be clear in advance whose task it will be to carry out these ranking and reconciliation exercises. What will happen to the EcoQOs which are ranked low or are incompatible?

As the number of EcoQOs increases, so does the risk of redundancy or, more seriously, mutual incompatibility. In attempting, for example, to restore commercial fish stocks, and fish and benthic communities to some improved state, the population dynamics for some seabird and marine mammal species maybe affected in such a way as to, at the very least, inhibit future population growth, if not cause actual population declines. In considering such potential conflicts, the logic behind the different objectives needs to be carefully maintained. The goals for commercial fish stocks and fish and benthos communities appear, at the very least, to be to return the system to a state characteristic of several decades ago. Some seabird species are currently at population sizes many times higher than they were at the start of the twentieth century. Much of this increase has been attributed to fishing activity: the provision of additional food resources at key times of the year through discarding, the increase in the abundance of small fish in the assemblage through size-selective fishing, and the removal of large predatory fish that may have competed with seabirds. Changes within the fish components of the ecosystem to a greater proportion of larger fish and fewer discards may render the North Sea a much more inhospitable place for some species of seabirds. Are EcoQOs for seabirds likely to reflect this, and allow for significant declines in some of our most abundant seabird species? Or will they be set so as to try and conserve the current state?

These difficulties are nearly unavoidable, if EcoQs for the ten EcoQ issues are developed and implemented independently. This decision may prove to have been pragmatic from the point of view that it by-passed the enormous hurdle of determining one (or at most a few) holistic ecosystem objectives, if such even exist, and so allowed the process to proceed quickly. However, the same hurdle may simply be encountered later, when it comes to putting the process into practice. At that point it will be necessary to gain social consensus on ranking which EcoQOs to pursue most aggressively, and on compromises to reconcile incompatible EcoQOs. Because these are human issues, clearly social scientists need to be more involved in the EcoQ and EcQO initiative.

To balance this pessimistic view, there are some potential steps forward. Short of the grail of one (or a very few) all-encompassing EcoQ and EcoQO, some simplification of the implementation task can be achieved by recognizing opportunities, if they exist, for one EcoQ to address more than one of the ten issues. This may be practical, regardless of whether one believes that a single well-chosen community-scale EcoQ may protect many species of fish, seabirds, marine mammals and benthos, or that an EcoQ for a well-chosen species, sensitive and vulnerable to several threats, may ensure the ecological quality of many other species and the larger community of which it is part. Also, a policy framework is developing that may guide ranking and reconciliation of EcoQs. The 1997 Intermediate Ministerial Meeting on fisheries laid down some guiding principles that require the development of an ecosystem approach to management, taking account of critical ecosystem processes, and involved a multispecies approach. This will be difficult or impossible to realize without giving priority to EcoQOs that are related to OSPAR’s communities and ecosystem process issues, even if they are difficult to make operational.

5.2 Ecosystem Properties and EcoQ Metrics

5.2.1 Background

The Convention on Biological Diversity (CBD), signed at the 1992 UNCED in Rio, provides the principal framework for international efforts to protect natural resources. The CBD defines biological diversity as “*the variability among living organisms from all sources including,* inter alia*, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems”*. This definition recognises, therefore, two components of biological diversity: the biological composition (itself divided into three levels—diversity among ecosystems and habitats, diversity of species within an ecosystem or habitat, and genetic variation within individual species) and the preservation of the ecological complexes of which they are part, that is to say ecological functionality.

The 1995 Jakarta Mandate on Marine and Coastal Biological Diversity (Conference of the Parties decision II/10) highlighted five (now six) thematic areas, second amongst them being sustainable use. The text specifically calls for “*the present mono-species approach to modelling and assessment should be augmented by an ecosystem process-oriented approach, based on research of ecosystem processes and functions, with an emphasis on identifying ecologically critical processes that consider the spatial dimension of these processes”*. The ecosystem approach is further defined (Conference of the Parties Decision V/6) as “*the application of appropriate scientific methodologies focused on levels of biological organisation, which encompass the essential structure, processes, functions and interactions among organisms and their environment. It recognises that humans, with their cultural diversity, are an integral component of many ecosystems”*. This again emphasises the need to consider not just protection of the full inventory of taxa present but also protection of ecological processes and explicitly the spatial elements of these processes.

This section provides major concerns to be addressed when specific EcoQs and their metrics are sought on each of the three key properties of the system.

5.2.2 Biological diversity

Most metrics of biological diversity can be derived from observations collected routinely during surveys. However, they never reflect the “true” diversity within the ecosystem, but rather the diversity as observed in the image of the community as viewed through the sampling gear. This picture is unavoidably distorted by species-specific differences in catchability, the absence of information on grounds that are difficult to sample, differential ease or vigilance in species identification, etc. If such metrics are used as an EcoQ, the inherent assumption is that any relative change in the survey metric greater than the sampling variance mirrors a true relative change in the ecosystem. In practice, surveys carried out with different methodologies (or different geographical extensions) may be expected to show differences in the same metric. Clearly, our ability to make conclusive statements about perceived changes in an EcoQ would be greatly enhanced if at least two independent surveys could be used to calculate the same metric. If these estimators would show similar annual deviations and trends, our confidence in measuring the true EcoQ of the system would obviously be increased.

5.2.3 Ecological functionality

Metrics of ecological functionality are even more problematic, because they can only be based on integrated sets of observations from different sampling programmes, each of which may be biased in specific ways. For many aspects of functionality, additional tropho-dynamic modelling is required to obtain the functional responses of the ecosystem and its components. Consequently, metrics of ecological functionality reflect modelling results rather than direct observations. In practice, any metric will be at least partly influenced by model assumptions even when model inputs are regularly updated with new observations, and the interpretation will often be open to scientific debate. Also, it is much more difficult to get independent confirmation, unless a suite of models with alternative assumptions is available and the robustness of model outcomes has been tested and found to be high.

5.2.4 Spatial integrity

Ecosystems may be defined at many spatial scales, but within the OSPAR context they apply to relatively large scales ("Large Marine Ecosystems"), that integrate over many sub-systems (pelagic vs. demersal; shallow vs. deep water; etc.). In fact, the spatial integrity of the different sub-systems could be viewed as an important element of total ecosystem quality. Spatial statistics are a specialized field (Ripley, 1988), and metrics derived from that field have not worked their way into most ecological practice. However, attention must be drawn to the fact that many metrics specifically apply to particular sub-systems (metrics derived from trawl surveys, for instance, provide specific information on demersal fish communities in muddy and sandy areas that can be trawled). Such restrictions may in fact favour the ability to assess some impacts on spatial integrity aspects of EcoQs. For example, changes in a metric directly related to the spatial impact of bottom trawling are effectively derived from the same suite of species as represented in the survey. If EcoQs and EcoQOs are to be effective tools for conservation of spatial integrity of ecosystems, however, an integrated and comprehensive set of quality metrics for spatial integrity are required, covering the entire suite of impacts caused by human activities.

5.2.5 Metrics

WGECO drew upon the group's collective experience to generate a list of key ecosystem properties relating to Biological Diversity, Ecosystem Functionality, and Spatial Integrity (Table 5.3.4.1). For each property, it went on to list at least a few key metrics. For some properties, there are very large numbers of possible metrics, often differing in only minor details. These lists are not exhaustive but cover examples of the most widely used metrics in each family. Nor did WGECO conduct an exhaustive critique of the relative merits of alternative metrics for various properties, a task which has been done many times before both by this group (ICES, 1995, 1996a), and in publications (see Hollowed *et al.*, 2000, for reviews; Rice, 2000). Rather, where possible WGECO chose metrics that were either in widespread use, or were recommended by recognized experts for certain fields of study, expecting that among equal alternatives, users of EcoQOs would prefer metrics with both of those features. Term of Reference e) for WGECO did not require that we complete all work on community-scale EcoQOs at this meeting. Therefore, WGECO thought that the most important task was to develop a rigorous and sound approach for identifying particularly promising or dangerous types of metrics at this meeting, identify important community properties for which promising metrics were not available, and refine the selection subsequently.

Aside from spatial integrity, WGECO has satisfied that no really major aspect of biological diversity or functional integrity would be missed by the properties and their metrics as tabulated in Section 5.3. WGECO also specifically considered and rejected calling the list of properties and their metrics either necessary or sufficient to, individually or in combination, ensure conservation of ecosystems, were they implemented in an EcoQ/EcoQO framework. Rather, each metric should be evaluated on its merits, with a watchful eye for redundancies, potential synergies, and gaps among promising metrics.

WGECO specifically assumes that high standards of quality control are applied at all stages of collecting data and conducting analyses to produce values on a metric (whether reference values or estimates of the present state of the system). Even the best metrics cannot withstand poor practice. Some metrics are especially vulnerable to distortion by even minor weakness in data sets or analysis approach, and such vulnerability must be considered when selecting metrics for use in reflecting EcoQs.

5.3 Evaluation

5.3.1 The evaluation method

In order to provide a unified framework for comparison of approaches, WGECO developed a cross-tabulation approach. We began by listing the ecological qualities that might be threatened by anthropogenic activities. These were considered in three categories: issues relating to biodiversity of species, to ecological functionality, and to spatial integrity of ecosystem properties. For each of these, we then listed a number of classes of metrics of that property. Each of these was then independently ranked by WGECO members against the eight criteria developed from those used by WGSE, WGMMPH, and Piet (2001) in the draft EcoQOs for fish (see Section 7.3). These criteria were designed to cover the utility of the metric both as an accurate measure, a property responsive to management action, and its communicability. All metrics that were considered could provide ecological information of great utility in the consideration of ecological dynamics and processes. For use as EcoQ metrics, however, the key issue was to determine which of the metrics at this time could form a basis for management given current levels of knowledge. In addition to selecting metrics, we also highlight areas where further metric development is required, either because no metric currently exists or because those available do not fully meet the criteria and so require additional development.

5.3.2 Criteria for good Ecological Quality metrics

The concept of ecological quality objectives (EcoQOs) has been discussed in a number of documents and at a number of recent meetings (Anon., 1999; Lanters *et al*., 1999; ICES 2001a, 2001b; Kabuta and Enserinck, 2000; Piet., 2001). Several key features of EcoQ metrics may be derived from these discussions. These may be summarised as follows:

Metrics 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.

These eight properties were all deemed desirable in a metric of EcoQ but were not all regarded as essential properties. The eighth was considered to refer to the information content of the metric rather than being a necessary quality. We therefore did not employ these criteria in our screening process.

5.3.3 Properties and metrics considered for fish and benthic communities

In the following annotated list, a number of properties of fish and benthic communities are reviewed and for each property one or more potential metrics are proposed. In all cases our assumption, in discussing a metric, is that it has been correctly calculated based on an appropriate data set.

5.3.3.1 Biodiversity of species

5.3.3.1.1 Biomass

Sum of weights across species from survey

The total biomass of organisms sampled, standardised for effort, from a region is an informative measure of its long-term productivity, and changes in long time series data sets show particularly useful broad scale change.

5.3.3.1.2 Size structure

Slope size-structure

Sheldon *et al.* (1972) showed a log-linear relationship between fish biomass and size. In spite of the differences in numbers and size between species, the community as a whole shows a log-linear decrease of biomass with increasing size. The slope of this relationship is assumed to reflect the efficiency of energy transfer and the mortality rate and can be used as a metric of the size-structure. Although several alternatives have been suggested since its introduction (Borgmann, 1987; Boudreau and Dickie, 1992; Boudreau *et. al.,* 1991; Thiebaux and Dickie, 1992, 1993; Sprules and Goyke, 1994), the conceptual basis is widely recognized (Rice and Gislason, 1996).

The general formula for the log-linear relationship between size and biomass is:

$$\ln(y) = a^* \ln(x) + b$$

where: x = size, y = biomass or number, a = slope, b = intercept.

A disadvantage is that slope and intercept are not independent, which makes it difficult to interpret a time series of either one. Also, an arbitrary choice must be made about the minimal size of fish that should be incorporated in the linear regression; depending on the mesh-size of the gear, certain size-classes will be under-represented and thus disturb the relationship.

Rice and Gislason (1996) studied the log-linear relationship for the North Sea fish community (1975–1995) and observed a change in slope caused by a decrease in large fish. This change was attributed to the impact of fisheries. Gislason and Lassen (1997) showed that a linear relationship between fishing effort and the slope of the size spectrum can be expected. WGECO (ICES, 1998) reported that there is now sufficient theoretical and empirical evidence to be confident that changes in fishing mortality should result in a long-term change in the slope of the size spectrum. Provided that growth and relative recruitment of the constituent species do not change, the change in the slope should be directly proportional to the change in exploitation rate of the community.

Length-frequency distribution

The length-frequency distribution of the community is determined by summing up the number of individuals caught per size class. In most cases these size classes will be cm-classes. A relevant metric to represent the length-frequency distribution may be the total number or weight of the community above a specific length threshold. Another relevant metric that may be derived from the length-frequency distribution is the percentage composition of groups that cover certain size ranges.

Multi-dimensional ordination

For studies involving complex tabular data (commonly i rows as sampling sites, j columns containing species or size-classes and cell entries of (transformed) abundances of species or size-class j at site i), ordination methods can be used to reduce this complexity to a small number of (usually) orthogonal (i.e., not correlated) gradients (reviewed in Jongman *et al*., 1987). Several ordination methods exist such as Principal Components Analysis (PCA), Correspondence Analysis (CA), and Non-metric Multidimensional Scaling (MDS). Of these methods, MDS has become the preferred technique for ecological ordinations of fish communities because of its increased robustness in the face of irregular distributions of abundance and high sampling variance (Clarke and Ainsworth, 1993; McRae *et al*., 1998). Although this technique may reveal patterns or trends that would otherwise remain obscured, interpretation or linking them to useful management information proves difficult, and communication to non-scientists challenging, to say the least. Although ordinations are listed under size structure, ordinations on the basis of species abundances as well as frequencies of size classes are common, so there could be ordinations of Species Identities.

5.3.3.1.3 Species identities

Species presence / abundance

There are several informative measures of community structure that do not take into account the species identities of the community. It is conceivable therefore that changes to species presence or absence may go undetected unless reference is made to lists of species relative abundance.

Index of rare species

Variability in abundance of the uncommon species in a survey can illustrate underlying patterns of change that are not evident from analysis of the dominant parts of the community. For example, the presence of unexpected migrants or the decline in population size of less common species can be used as metrics of previously unobserved adverse human impact. Daan (2001) proposed a spatial and temporal diversity index that was based on species rarity.

Index of declining or increasing species

A variety of metrics are available based on the proportion of species in the community which are showing increases or decreases in abundance (biomass). These measures are at best coarse and may provide little information about causes of the changes, but are readily interpreted and understood by non-specialists.

Presence of indicator, charismatic, sensitive species

Societal concerns about the environment often focus on a limited number of organisms that are in some way “attractive”. Such charismatic species, including dolphins, killer whales, large sharks, and a variety of seabirds, are often viewed as sentinels of the health of the ecosystem. The scientific justification for such a view varies with the species, but as many are higher predators and long-lived they will often be more sensitive to human impacts. Indicator and sensitive species are selected on the grounds of criteria that explicitly use their known response to impacts. Many examples of such indicator taxa exist in the pollution literature (Pearson and Rosenberg, 1978) and a limited number of benthic taxa have also been suggested as being vulnerable to direct effects of fishing (Lindeboom and de Groot, 1997). Development of this approach is often more difficult than it at first appears as lists of sensitive/indicator taxa are rarely transferable between regions and developing the list from the impacted system studied leads to circularity.

Non-indigenous species

The presence of non-indigenous species, used here to mean species introduced by anthropogenic activities rather than natural invasions/range expansions, is by definition a failure to maintain “natural levels of biological diversity”. For larger organisms, the presence of non-indigenous species is easily recorded; for lower organisms, our lack of knowledge of pristine fauna makes this more difficult (Eno *et al.*, 1997).

Species turnover/loss rates

The rate at which species composition changes from year to year in samples taken in a consistent manner and location is a widely used metric in terrestrial conservation biology. It requires consistent and reliable sampling where sampling is expected to detect most of the species that are present. Measures of turnover rates are most effective at local scales, and may be less effective at the scales of large marine ecosystems when many samples are pooled.

5.3.3.1.4 Species diversity

The concept of species diversity has a long history in the ecological literature; countless different metrics have been devised and utilised in numerous different studies covering taxa from just about every phylum in the plant and animal kingdoms (Brown, 1973; Connell, 1978; Davidson, 1977; Death and Winterbourn, 1995; Eadie and Keast, 1984; Heip *et al.*, 1992; Huston, 1994; MacArthur and MacArther, 1961; Magurran, 1988; May, 1975; Rosenzweig, 1995; Washington, 1984). Despite this long tradition, and perhaps in part due to the proliferation of different metrics, species diversity as a concept has been questioned (Hurlbert, 1971). Hill (1973), however, argued that much of the perceived difficulty with the concept lay in the fact that it combined the two characteristics of richness and evenness. The theoretical underpinning of the concept has been discussed (May, 1975, 1976). The ability of the different indices to actually detect environmental and anthropogenic influences has on occasion been questioned (e.g., Robinson and Sandgren, 1984; Chadwick and Canton, 1984), however, in general these problems have usually been associated with inadequate sample size (Soetaert and Heip, 1990).

WGECO considered several species diversity metrics as candidates on which EcoQOs could be based. The simplest representation of the species relative abundance data, on which any metric of species diversity is based, is the straightforward graphical representation of relative abundance on species abundance ranking. The most commonly used representation of this type is the *k*-dominance curve (Lambshead *et al*., 1983; Clarke, 1990). This index was endorsed by WGECO because of the simple, easily comprehensible way that it conveyed the information, avoiding the problems of trying to convey both aspects of species diversity in a single numeric parameter. Well-defined statistical methods for determining differences between samples have been developed (Clarke, 1990). The *k*-dominance curve was the only metric to receive a positive score for all selection criteria.

Hill's N numbers

Hill (1973) suggested that several of the most commonly used diversity indices were mathematically related, forming a family of indices varying in their sensitivity to species richness and species evenness (Peet, 1974; Southwood, 1978). These indices are all affected by sample size, which is a major disadvantage with regard to monitoring change in marine ecosystems where sampling is logistically difficult and expensive. As the Hill number notation increases, the index moves from being a measure of species richness to one of species dominance. Low N number metrics, e.g., N0 and N1, are consequently the most affected by variation in sample size. When the problem of variable sample size can be addressed, these metrics have been used to demonstrate long-term temporal and spatial trends in species diversity that have been associated with differences in fishing activity (Greenstreet and Hall, 1996; Greenstreet *et al*., 1999).

Taxonomic Diversity Indices

Taxonomic diversity indices were developed by Warwick and Clarke (1995, 1998). They are closely related to the Shannon-Weiner Index, but they also provide additional information with respect to the level of phylo-genetic relationship present in samples. As such they were considered to convey some information on the genetic diversity aspect of biological diversity. They have been demonstrated to be relatively sample-size independent, and to be sensitive to ecological perturbation in circumstances where other species diversity metrics, such as the Shannon-Weiner, or Simpson's Indices, fail to respond. They are, for example, particularly sensitive to situations where a group of particularly vulnerable, closely related species may be in decline and being replaced by alternative, unrelated species. The impact of fishing on elasmobranch fish species is an example of this (Rogers *et al*., 1999). However, in circumstances where Hill's N1 and N2 are varying, these taxonomic indices may convey little additional information (Hall and Greenstreet, 1998).

Theoretical Distribution Metrics

Log-Series and Log-Normal: Parameters derived from these distributions have the advantage of being relatively sample-size independent (Kempton and Taylor, 1974). Also, there has been considerable debate in the ecological literature regarding the theoretical reasons as to why distributions of species relative abundance should follow either one of these models (Fisher *et al*., 1943; Preston 1962, 1980; Kempton and Taylor, 1974; May, 1976). One major difficulty with using these indices lies in the necessity to fit the data to the distributions, to estimate parameters of the distribution for subsequent use. Generally this tends to require a substantial amount of data, rather negating the advantage of sample-size independence. Often fitting the data to the distribution proves to be difficult, and in testing the significance of any fit, one hopes not to disprove the null-hypothesis, which is unsatisfactory from a statistical perspective.

Species-Effort Index

Many scientists have argued on theoretical grounds that species richness (e.g., N0) is the most important aspect of species diversity, but the sampling effort required to estimate this adequately from the data normally available from fish or benthic surveys is usually prohibitive. WGECO considered that a species-effort index derived from the parameters of the function describing the rate of increase in the number of species recorded as samples from a survey are increasingly aggregated may offer a solution. This function is exactly equivalent to the species-area relationships of the form $S=cA^z$, which describes species richness in habitats of varying size, e.g., islands, continents (Rosenzweig, 1995). The two parameters, c and z, could perhaps be derived from a much smaller number of trawl samples to provide a relatively sample-size independent estimate of species richness.

5.3.3.1.5 Life history composition

There is extensive theoretical literature that distinguishes K-strategists from r-strategists, that is, species whose life history characteristics adapt them to living in undisturbed, stable environments vs. those adapted to living in frequently disturbed, variable environments. Particular life history characteristics can be used to place species somewhere along this continuum, and thus provide an indication of vulnerability to disturbance by additional fishing mortality. Correspondingly, the life history character composition of communities may provide a metric of the past impact of fisheries on that community. Possible life history characteristics that might be used as such metrics include:

- maximum size (cm);
- size above which 50 % of the population is mature (cm);
- maximum age (year);
- age above which 50 % of the population is mature (year);
- fecundity expressed as number of eggs per female or number of eggs per body weight;
- parameters k and $L\infty$ of von Berthalanffy growth curve.

Values for one or more of the above parameters are available for many species from the literature. This list, however, is far from comprehensive and for several of the parameters values are available for only a few species. Community metrics based on these parameters are calculated per year by weighting the community species' biomasses with the value of that particular life history parameter.

Other potential metrics might be derived from sex ratio, lifetime reproductive output, or growth rates.

5.3.3.2 Ecological functionality

5.3.3.2.1 Resilience

The concept of resilience refers to food webs as a whole (Pimm, 1982; Cohen *et al.*, 1990). The concept addresses the ability of the web as a whole to retain its overall configuration when stressed, or to return to its original configuration when perturbed. Food webs can suffer several types of stresses and perturbations, including invasions by new species, loss (extinction) of species in the web, and large, abrupt increases or decreases in abundances of one or more species. There is much theoretical detail about what properties of food webs do (or do not) make food webs (and the ecosystem that they represent) amplify or damp stresses and perturbations, and about what constitutes an important response by the food web. For use as a general metric of food web (ecosystem) quality, however, the diverse expert argumentation consistently suggests that "healthy" food webs (ecosystems) maintain their general configuration when moderately stressed or perturbed, whereas badly altered ones may undergo dramatic restructurings by the same degree of stress or perturbation. There are, of course, the usual problems with potential circularity of the concept, and concerns that Null Hypotheses are often poorly formed when the concept has been tested with models or in the field. Theory about **resilience** of food webs has identified a number of potential metrics. The ones considered by WCECO include:

Return time of properties of food webs

This refers to the number of time steps required by a food web to return to its original configuration when perturbed in some specified way. Stable food webs should have short return times, and return times increase as food webs lose properties that confer stability. The parameter for which return time is measured depends on the model or study, and selection of the parameter can affect the results. If the metric is used as a measure of ecological quality, it is also necessary to decide whether the state to which the food web (ecosystem) should return is a recent state, or a state thought to persist historically.

Invasibility

The likelihood that a new species can establish itself if introduced into an existing food web. Sometimes the measure differentiates cases where a successful invader can be established without loss of any species in the original food web. At other times the measure includes the degree to which membership of the previous web was changed by a successful invader. Invasibility depends, of course, on the characteristics of the "species" introduced, so this property is usually explored through intensive simulations. Such simulations have demonstrated that some configurations of food webs are more likely to allow invading species to be established than others, and some configurations of food webs are more likely to lose existing species when an invader is established that others. Field studies sometimes have confirmed predictions from theory, and other times have not. It is generally argued that as communities co-adapt to particular environmental conditions, invasibility of food webs should decline, and when food webs are stressed invasibility may increase.

5.3.3.2.2 Productivity

Although there are many ways to measure productivity, the basic concept is the amount of new material produced by some level of biological organization. Productivity has been discussed sometimes at the scales of individual (growth), but more generally at the scale of species (increase in numbers and/or biomass), and ecosystems. At the scale of ecosystems, primary productivity (fixation of carbon by plants) is generally differentiated from secondary productivity (passage of carbon [or other currency] through the food web). System productivity is also often partitioned into "new" production, due to nutrients taken from inorganic sources, and "regenerated" production, due to recycling nutrients already in the food web. There is again much theoretical detail in this area (Cushing, 1995; Steele, 1998). In the context of maintaining ecological quality, however, the property is considered quite broadly. Ecosystems that are highly productive, producing lots of biomass, energy, and/or individuals are considered to be in "good" condition with high ecosystem quality (unless excessive nutrient inputs cause eutrophication). As the quality of the ecosystem (or any of its components) is degraded, its productivity can decrease, and less "stuff" is produced.

Secondary production occurs in the water column (zooplankton) and on the seabed (benthos). On-site measurements of secondary production in the North Sea of all seabed animals have not been made, also due to the lack of adequate methods. Only sporadic measurements have been executed into the secondary production of specific species. The fish community in the North Sea is situated on the third and fourth trophic levels and as such is dependent on the production of the underlying levels. The total fish production can best be determined based upon stock assessments of all the fishes occurring in the North Sea. However, stock assessments have only been made of a number of commercially important species, but they do form a significant share of the total fish biomass. An estimation of the total fish production is the sum of the somatic fish production and the production of gonads.

P/B ratio

The ratio of production of some part of an ecosystem to the standing biomass of the same part of the ecosystem. This can be measured for a population, a suite of species, a trophic level, or any other grouping that researchers can quantify and justify.

Carbon per unit area/time/volume

In general, productivity is expressed as the fixation of amount of carbon per area per time unit (e.g., a regular expression for primary production is for instance g C per m^2 per year).

Partitioning of production between somatic and gonad material

This in effect follows on from the discussion on life history characteristics above. As the community shifts towards domination by r-strategist species, the partitioning of production between gonadal tissue and somatic tissue should shift from investment in somatic material to investment in gametes. This follows on from the nature of the two types of strategists. K-strategists invest in growth because they intend to remain for a long time in a stable home. Conversely, r-strategists tend to have small body sizes. Instead, they mature early so that, from that point, they cease investing heavily in growth, directing their resources to producing gametes instead. This buffers them from perturbation in the environment, ensuring that they can recolonise an area, or colonise an alternative area. Consequently, in a community disturbed by fishing, one might expect a shift in the ratio of gamete:somatic production.

5.3.3.2.3 Trophic structure

Trophic structure is a general term for the feeding relationships among species in a community and ecosystem. Theory on trophic structure has a long history and can be quite complex (Pimm, 1982; Cohen *et al.,* 1990; Hall and Raffaelli, 1991; Rice, 1995; Thingstad, 1998). In general, however, trophic structure is thought to be a major component of how communities and ecosystems maintain their integrity. Abundance of individual species within a trophic system may change due to human perturbations, environmental forcing, or the trophic (predator-prey) relationships themselves. The trophic structure is some consolidated or emergent statement about how the relationships among the species respond to those changes in abundance, whether tracking them proportionately, amplifying them, or buffering them. Trophic structure is often expressed for aggregates of species, often grouping species into levels sharing a common number of trophic transfers: primary producers being the first level, their grazers being a second level, predators on grazers being a third level, etc. Because feeding is strongly size dependent in marine ecosystems (see Size Structure), these groupings are generally severe abstractions of reality. Nonetheless, they form the basis for most analyses of trophic structure.

By representing the relationships among predators and prey, trophic structure is considered fundamental to ecosystem functioning. Human actions that alter trophic structure are generally considered to degrade ecosystem quality, particularly if the change simplifies the structure in some way, such as reducing linkages among species or the proportion of total biomass at any level.

Distribution of production among trophic levels, size classes, taxonomic groups

This represents a class of metrics that are simply the frequency distribution of productivity (measured as biomass, calories, etc.) across a number of groups of species to another, where the grouping criterion could be trophic level, size classes, etc.

Connectance

The connectance index in a food web is the ratio of the number of actual predator-prey links to the maximum number of possible links, where different modellers have applied slightly different approaches to determining the theoretical upper limit. Christensen *et al*. (2000), for example, estimated the number of possible links as $(N - 1)^2$, where N is the number of food web groups.

Path length

This is a measure of the distance, measured as number of linkages, between selected species (or nodes, if species are aggregated in a food web model). Different researchers have used the mean number across all linked species, or the distance from primary producers to top predators, as the maximum number of steps possible in a model as the metric for estimating the path length of a food web. Christensen *et al.* (2000) estimated path length as the average number of groups that an inflow or outflow passes through in their models.

Ratios of trophic levels

This represents a class of metrics that are simply the ratio of biomass or productivity (measured as biomass, calories, etc.) of group of one species to another, where the grouping criterion could be trophic level, size classes, etc. There are as many possible metrics of this property as there are ways to group species and things which reflect their role in the ecosystem. Intended usage, data availability, and professional experience will guide the selection of grouping criteria and things to express as ratios.

5.3.3.2.4 Throughput

This property reflects the rate at which energy or biomass is passed through the ecosystem. It is influenced by ecological efficiencies of the species in the web, the numbers of linkages among species, and mortality rates. It is an important property of ecosystems, but to use it would require data not likely to be available without significant preparatory work, and probably much new directed research. Therefore, WGECO did not give prominence to metrics of it, such as:

Internal consumption to yield

The ratio of energy lost to the system through respiration and bioenergetic needs of the individuals in the web to the energy removed by the fishery.

Ulanowicz index

In his textbook on bioenergetic ecological models, Ulanowicz (1997) has a specific index that reflects throughput of energy in a food web. The Working Group was aware of the index, but lacking energetics data and the Ulanowicz book in the ICES library, this metric was not pursued.

5.3.3.2.5 Body well-being

Condition factor

In fish ecology, condition is believed to be a good metric of the general "well-being" or "fitness" of the population under consideration (Adams and McLean, 1985). This can also be expected to apply at the level of the community. Several condition indices are used in fishery science as metrics of the length-weight relationship of a population. However, the conversion of a two-dimensional length-weight relationship into a single statistic results in a loss of information and, in many cases, an inaccurate representation of that relationship. After review of the most common condition indices by Bolger and Connoly (1989), Cone (1989) propagated the calculation of estimates of ordinary least squares regression parameters as the most accurate method of examining length-weight relationships for fish populations. However, since regression parameters are commonly heterogeneous and slope and intercept are often inversely related, valid interpretation of the results is difficult (Bolger and Connoly, 1989). A disadvantage of an alternative, the estimated weights of fish of a particular species and length from regression equations specific to the groups under consideration (De Silva, 1985), is the dependency on the arbitrary choice of the length.

For the community, one possibility would be to use the average condition of a theoretical community of fixed size-structure and species composition over time as an index of body condition. For each individual in this community, the condition is expressed as the weight calculated from the species-specific length-weight relationship per year and the mid-range length of the size-class. Considering that length-weight relationships are only determined annually for a subset of (commercial) species, this theoretical community will consist of a subset of species that are present in the actual community. Another possibility would be to use the full frequency distribution of condition factors (calculated correctly) across a suite of species, and compare the distributions themselves across space or time, or compare their ordinations.

Incidence of disease, pathogens, parasites, contaminants

Considerations relating to the types and incidence of diseases and parasites are similar to those relating to body burdens of contaminants and other measures of body condition. If lower environmental quality affects the biological health of individuals, their resistance to disease and parasites may be lowered. Hence, it is possible that metrics based on the incidence of disease or parasites across a full community could be developed. Such a metric would require data not available to this meeting (and possibly not at all) and hence it was not explored at this meeting.

5.3.3.3 Spatial integrity

No specific metrics were identified for this property (see Sections 4.2 and 5.3), but the property was scored during the evaluation process.

5.3.4 Results of the evaluation

The resulting scores were discussed and Table 5.3.4.1 represents the consolidated results of this consensus building phase. Metrics were graded on a three-point scale: 2 fully matched to criteria, 1 of some utility against these criteria, and 0 fails to address at least some aspect of these criteria. As in any exercise of this nature, there were some areas of divergent opinion and a number of concerns that are summarised below.

In the tables evaluated, there were 320 cells and complete unanimity of scores was achieved in 30 % of the cells (95/320) for fish and 40 % (127/320) for benthos. WGECO then proceeded to remove all metrics that had been scored unanimously with a zero for any of the first seven criteria. It had been decided *a priori* that the first seven criteria were to be of equal weight, while the eighth was considered to refer to the information content of the EcoQ rather than a necessary quality.

This first selection left 21 measures in the fish matrix and 14 in the benthos matrix. This was still considered to be too many to be of use operationally and a second sifting was applied. We now removed all metrics with had a *modal* score of zero for any of the first seven criteria. This restricted the list for fish to seven metrics, although three cover one property (size structure) (Table 5.3.4.1). For the benthos only one measure, presence of sensitive/charismatic/indicator taxa, remained (Table 5.3.4.1).

WGECO then proceeded to consider if these strict criteria had excluded any metrics that tracked crucial properties and almost met the selection criteria (Sections 6 and 7) and to develop recommendations where key ecological qualities had no metric (Section 5.3.6). From this, we developed our recommendations in relation to ToRs b), c) and e) (Sections 6 and 7).

5.3.5 Metrics not considered further

In this section, we add some brief commentary identifying the principal reasons why various metrics were not considered further (i.e., the criteria they failed to meet).

Biomass: Total sample biomass did not meet the criteria as it was generally regarded as being insensitive to human impacts and subject to high levels of "noise" (natural variation) and for benthos there is a lack of historical data at the appropriate scale.

Size structure: Percentage size composition was the only metric of this group dropped from the fish table. It was considered to have lower sensitivity to human impacts than the other measures and so was dropped. In the benthic table all the metrics failed to meet the selection criteria primarily due to lack of existing data, confounding effects of sampling protocols, and communicability.

Species identities: Indices failing to meet the criteria in this category were generally regarded too insensitive to human impacts and subject to high levels of "noise" (natural variation).

Species diversity: The excluded metrics tended to fail on the criteria of "a high response to the signal from human activity". Many of these metrics are affected by environmental variability. Problems of sample size variability also tend to mask the signal. There was also concern that the "linkage in time" of many of these metrics was poor. Lag-times were too long, so that the delay between event and response was such that managers may not be able to take remedial action quickly enough. The theoretical linkage between fishing activity and diversity is also poorly understood. How does fishing affect species diversity, and exactly what type of change in activity is required to achieve a particular response?

Life history composition: A variety of life history metrics were considered and most were rejected for fish and all for benthos. The principal reasons were the extent of noise in the data, lack of a tight effect-to-response relationship, and difficulties of having sufficient data for assessing them.

Ecological Functionality: A host of metrics were reviewed for the properties considered relevant to ecological functionality. Productivity was the strongest candidate metric in this group, but all failed to meet the criteria. The main reasons for failure were difficulties in accurately measuring (deriving) the values, a particular concern for those only derivable from models, lack of a strong response to human effects, and a lack of historical values. These issues are addressed further in Section 5.3.6.

Spatial Integrity: WGECO was unable to propose a metric which adequately addressed this issue. There was considerable consensus that this was an important issue and it is considered further in Section 5.3.6.

5.3.6 Gaps

5.3.6.1 Metrics of biological diversity

Much of the reasoning behind the OSPAR EcoQ issues and the development of EcoQOs is driven by the commitments made by most European governments and the EC to the Rio Convention on Biological Diversity. Most biological sampling programmes undertaken in the North Sea invariably record information on species identity and abundance. Therefore, it should be within the power of fisheries scientists and marine ecologists to say rather a lot about species diversity. However, no single diversity index survived the criteria for the selection of metrics on which EcoQOs could be based. This highlights a major failing of the currently available range of diversity measures as operational metrics in the opinion of WGECO.

There is an extensive literature on the subject of species diversity, including theoretical and applied studies (Section 5.3.3.1). These studies have identified a number of shortcomings of the indices that are relevant to their use as management tools and triggers. Diversity indices encapsulate two characteristics of species relative abundance: the number of species and the distribution of individuals among species. Thus when the value of an index changes, it is rarely clear what has happened without further investigation. Species diversity indices vary considerably from year to year, so the signal-to-noise ratio is often low. Most of the metrics in use are sensitive to sample size and to vigilance of observation, weakening further the signal-to-noise ratio. In addition, the relationship between fishing and the species diversity of benthic and fish communities in the North Sea is poorly understood. For example, both positive and negative responses of diversity to fishing have been found (Greenstreet *et al*., 1999; Rogers and Ellis, 2000; Piet, 2001). Therefore, it would not be possible to advise managers of the adjustments to fishing effort that would move a diversity index towards a chosen value.

It would be inappropriate to suggest that any particular species diversity metric would provide an adequate metric of EcoQ in this respect, or therefore provide a sound basis for an EcoQO. Nevertheless, species diversity remains an important characteristic of the communities that make up the North Sea ecosystem, and work should be done to develop metrics of species diversity free from these shortcomings.

5.3.6.2 Metrics of ecological functionality

OSPAR and associated participants in the efforts to implement ecosystem-based management in the North Sea (and elsewhere) have made commitments to conserve ecological functionality as well as biological diversity (Section 5.1.3). WGECO supports this conceptual commitment, but found almost no metrics that could meet reasonable standards for use in management applications at present, nor were any of the ones considered by WGECO thought likely to meet them in the near future. This is a major gap, which requires both some explanation and constructive suggestions for making progress. WGECO identified three aspects of ecological functionality that could be considered separately. For each aspect, the prospects for development of community metrics were different.

1) *The well-being of the all the individuals in the community, when viewed collectively*. The community-wide distribution of biological condition has been designated as a promising metric, but no similar metrics were identified for community-wide distributions of body burdens of contaminants or incidence of diseases, parasites, etc. WGECO does view the community-wide level of contaminants, disease, etc., and how concentrations or incidence vary among species and individuals within a community to be an important attribute of ecological quality of the community, particularly when biomagnification and bioaccumulation compound risk or impede rehabilitation. Nonetheless, that does not mean that there is some community-scale metric of level of contaminant that would be more sensitive to perturbation or more informative to managers than contaminant levels or disease incidence in well-chosen indicator species. ICES has previously provided advice on selection of indicator species for contaminants (ICES, 1989). The only addition to the past advice on selection of indicator species when one is advising on community-scale indicators of contaminants or disease is the representativeness of the species being used. At the community scale, species which are widespread and highly mobile within the ecosystem of concern should accumulate contaminant and disease burdens more representative of the “community” than a species that is sedentary and patchily distributed, so contaminant levels reflect quite local conditions.

2) *The responses of biological processes to physical forcing*. Great strides have been made in linking physical oceanography to dynamics of marine populations and communities, especially processes like recruitment and growth in fish stocks (Harrison and Parsons, 2000; McKinnell *et al*., in press; ICES, 2000a). An ecosystem approach should take these linkages into account as fully as possible. Such considerations do not create the need for new community-scale EcoQs and EcoQOs for management, however. In general, the same metrics currently used in single-species fisheries management, for example, can continue to be used. What changes is that the estimation of current states of the population, projections of states in the near- and medium-term future, and possibly even the values of the reference points used in advice can all be improved. Some research on oceanographic forcing of biological systems is indicating that ecosystems may undergo relatively abrupt regime shifts (Francis *et al*., 1998; Reid *et al*., 2001), which could affect properties like productivity and resilience of the full system. It is not yet known how to accommodate fully regime shifts in single-species reference-point based advice and management. However, there is no reason to expect that the setting of some EcoQ and fixed EcoQO for a community property will be an effective strategy for bringing regime shifts into ecosystem management. Such a strategy has the risk of making management less responsive to oceanographic regimes, if they are important, rather than more responsive, by giving special status to some historic configuration of the ecosystem, instead of considering the ecosystem quality objective best for each regime.

3) *Tropho-dynamic processes*. These are intrinsically dynamic relationships among organisms, species and their environments and habitats. This stands in contrast to biological diversity, which is more of a structural property and, although dynamic over time, has a meaning when considered statically at a moment in time (or a sampling interval). Given that tropho-dynamic relationships only have meaning dynamically, they are less tractable to direct monitoring, and tropho-dynamic models are virtually essential in calculating values of metrics. Tropho-dynamic modelling has been an active science field for some years (reviewed in Hollowed *et al*., 2000), and WGECO has been following the field closely (ICES, 2000a). There is no shortage of tropho-dynamic models, and for over a decade ICES has been using multispecies models of predator-prey interactions (ICES, 1996b) as a contribution to the basis for scientific advice on fisheries. However, ICES has intentionally used these multispecies models to improve estimates of specific parameters of assessment models, with advice continuing to be based on single-species properties that are again estimated better.

With regard to integrated properties of the multispecies models, ICES has viewed results as matters of research interest and tools for framing ecological hypotheses (ICES, 1988, 1990), but not as suitable bases for management advice. When considering tropho-dynamic models of even greater portions of ecosystems, WGECO sees no reason to change its past conclusion (ICES, 1998, 2000a) that none are presently suitable to use as the basis for management advice. Various tropho-dynamic models can produce many outputs that may appeal as bases for advice, but the appeal is deceptive. Tropho-dynamic ecosystem models are still research tools at best. They have not been tested with the rigour routinely applied to models that are used by ICES in formulating management advice, nor do the data used in parameterization withstand the review given to data accepted for analyses by most ICES Working Groups.

Many things have to improve before ecosystem tropho-dynamic models should be viewed as suitable sources for advice on specific management problems, for use in setting EcoQOs. Databases of feeding relationships of marine predators have to cover many more species in the ecosystem, and must be updated on time scales at least matching the time scale on which advice is required regarding properties derived from the tropho-dynamic relationships. Data on energetic requirements of predators and energetic values of prey, and how they vary in space, time, with size, and with abundance of other species are often even weaker than the diet data, and require even more augmentation. Better data for parameterization will not be sufficient for tropho-dynamic models to be suitable for use in advisory contexts, however. The models themselves have to be improved through addition of important processes, such as environmental forcing of system dynamics (see above) and food-dependent life history dynamics (Pimm and Rice, 1988; Rice, 1995), and effective treatment of uncertainty about data and formulations of relationships. More importantly, the models have to undergo a level of testing and validation with a much greater rigour than has been customary when the models are used in exploratory modes. The workshop on testing ecosystem models that was recommended last year (ICES, 2000a) is an important step in the right direction. However, based on the results of the Planning Group meeting (ICES, 2001c), it appears that the existing ecosystem models are still far from being amenable to the type of testing necessary for their use as a basis for management advice, let alone being ready to pass such tests.

With a continuing pessimistic view of the value of ecosystem models to improve management advice directly, particularly in terms of providing currency for effective and reliable EcoQOs, WGECO may be becoming perceived as de-emphasising the importance of tropho-dynamics in understanding ecosystem processes, and in making management of marine ecosystems truly effective. Rather, the opposite is the case. WGECO considers these relationships very important for conserving ecosystem functionality, and certainly sufficiently important that models of the relationships need to be tested as rigorously as models used for the comparatively much simpler problems of tracking, forecasting, providing information about, and supporting scientific advice on single-species dynamics. These are new challenges to ecosystem modellers, but challenges they must rise to meet, if tropho-dynamic aspects of ecosystem functionality are to be convertible into EcoQOs.

It will take time to meet these challenges, and for the interim, it may be more effective to look at much simpler attributes as candidates to be surrogate metrics of tropho-dynamics aspects of ecosystem functionality. WGECO noted that things as simple as the mean and distribution of mouth-gape sizes of predators might be informative about tropho-dynamics at the community scale. This trait, and other similar traits, should be explored in the context of a possible metric for use in ecosystem management, while the longer-term work on raising both tropho-dynamic ecosystem models and their testing approaches to another plane of rigour and reliability, is pursued.

5.3.6.3 Metrics of spatial integrity

Several statistical measures of spatial pattern exist (e.g., Ripley, 1988) and there are many measures used by researchers in studies of spatial structure of populations and meta-population dynamics (Cooper and Mangel, 1999; Caroll and Lamberson, 1999; Policansky and Magnussen, 1998). Between these two sources, there would be no shortage of metrics that address in some way ecological issues of spatial structure and/or function. This does not mean that the possible metrics are good candidate metrics for **community**-scale measures of **spatial integrity. First**, many of them have only been used in single-species applications, and even their computation at the scale of a community may not be straightforward, or possible at all. Where it turns out to be possible to compute the metrics of spatial pattern or meta-population relationships at the community scale, the ecological interpretability of the results remains to be established. **Second,** to the knowledge of the Working Group, the usefulness of most of the spatial metrics has not been tested and demonstrated to be effective in management contexts. This does not mean that we believe that they are not useful when advising managers, simply that their hit, miss, and false alarm rates in management applications are largely unknown. Nor for most or all metrics will their linkage to management actions and their time sensitivities to perturbations be known. **Third**, even if there are metrics of spatial pattern or meta-population relationships that are computable and applicable in management contexts, it is far from clear how to know the degree to which the metric(s) reflect the fairly abstract property of **spatial integrity.**

Is there a reason to be concerned about the absence of community-scale metrics of spatial integrity? Spatial pattern, particularly habitat fragmentation, is a dominant concern in the management of many terrestrial and coastal ecosystems (Eggleston *et al.*, 1999; Olsen, 1999). In marine ecosystems, it should be much less of a concern, because larval distribution processes for many marine fish and invertebrates spread eggs and larvae very widely in the ecosystem. We stress that this is not an absolute exemption from concern, however, because recruitment processes of some important marine plants such as eelgrass may be very local, and some marine invertebrates such as dogwhelks also spread very slowly. Particularly where plants constitute an important part of the marine habitat, spatial integrity may be an important consideration. Also aside from recruitment processes, spatial relationships may be crucial to interactions among predators, prey, and competitors (Rothschild and Osborn, 1988).

Not only are there ecological reasons to conclude that spatial pattern/integrity contributes to ecological quality, there are management issues with intrinsic spatial components. The design of marine protected areas to achieve biological and conservation objectives should be informed by EcoQOs reflecting spatial integrity, if any could be developed. Although WGECO has stressed many times that reducing fishing effort is an essential step to reducing impacts of fishing (ICES, 2000a), for a given amount of fishing effort, changing the spatial pattern of fishing may contribute to changing ecological quality. This would again give value to informative measures of spatial integrity, were any to be found. Finally, a number of coastal zone management issues have an inherently spatial component, and informative metrics of spatial integrity could again be helpful in managing for improved ecological quality.

If there are ecological and management reasons to be interested in metrics and EcoQOs for spatial integrity, what should be done to rectify their present absence? First, members of WGECO must familiarize themselves more fully with the research field and literature on spatial statistics and meta-population dynamics, and must attract a few experts to the next meeting where this topic is addressed. Advances from the growing field of landscape ecology (Kareiva and Wennergren, 1995; Gray, 1997), to this point pursued largely for terrestrial systems, also need to be brought into marine applications as focused research and not vague platitudes. Knowing more about the ecological information in spatial metrics whose operational management relevance has not been explored, will be only a small step forward. It is critically important that the functional utility of these metrics to support management decision-making also be explored in a focused way. This will require new types of research on these metrics, as discussed below.

Table 5.3.4.1. WGECO group grading of various ecosystem metrics for properties covering key ecological qualities. Metrics were graded on a three-point scale: 2 fully matched to criterion, 1 of some utility against this criterion, and 0 fails to address at least some aspect of this criterion. See the text for a description of the metrics and justification for the criteria: (a) species biodiversity fish communities, (b) species biodiversity benthic communities, (c) ecological functionality in general, and (d) spatial integrity. Where there was unanimity in grading, a single value is presented; otherwise the range of scores is given.

Properties	**Possible metrics**	**Comprehen-sive and com-municable**	**Sensitive to manageable human activity**	**Tight linkage in time to that activity**	**Easily and accurately measured**	**High response to signal from human activity compared with variation induced by other factors / low miss rate**	**Measurable in a large proportion of the area to which the EcoQO is to apply**	**Measured over enough years to provide baseline of information and allow realistic setting of objectives**	**Representative of relevant aspect of EcoQ. May relate to wider environmental condition**
A. SPECIES BIODIVERSITY FISH COMMUNITY									
Biomass									
Size structure	Slope of size spectrum	0–1	1–2	0–1	2	0–2	2	2	1–2
	Length frequency distribution	0–2	1–2	0–2	1–2	0–1	2	1–2	2
	Mean length/weight of all organisms sampled	1–2	1–2	1	2	0–1	2	2	2
Species identity	Presence of indicator, charismatic, sensitive species	1–2	1–2	0–1	1–2	1–2	1–2	1	2
Species diversity	k-dominance curves	1	1–2	0–1	1–2	0–1	1–2	1–2	2
Life history Comp	L_{max} (weighted mean, full distribution)	0–2	1–2	0–1	1–2	1	2	1–2	2

Properties	Possible metrics	Comprehen-sive and com-municable	Sensitive to manageable human activity	Tight linkage in time to that activity	Easily and accurately measured	High response to signal from human activity compared with variation induced by other factors / low miss rate	Measurable in a large proportion of the area to which the EcoQO is to apply	Measured over enough years to provide baseline of information and allow realistic setting of objectives	Representative of relevant aspect of EcoQ. May relate to wider environmental condition
B. SPECIES BIODIVERSITY BENTHOS									
Biomass									
Size structure									
Species identity	Presence of indicator, charismatic, sensitive species	1–2	1–2	0–2	1–2	1–2	1–2	1	2
Species diversity									
Life history Comp									
C. ECOLOGICAL FUNCTIONALITY									
Resilience									
Productivity									
Trophic structure									
Throughput									
Body well-being	Mean and distribution of body burden (contaminants)	1–2	1–2	1–2	1–2	1–2	1–2	0–2	2

D. SPATIAL INTEGRITY

5.4 Framework considerations

The ICES approach to fisheries advice and the OSPAR approach to ecosystem management differ because OSPAR focuses on one goal, achieving a desired state of the Ecological Quality Objective. The OSPAR approach gives no role to limit and precautionary reference points, which ICES defines relative to undesirable states to be avoided with high probability. The ICES approach includes explicit provisions for uncertainties from several sources, whereas the OSPAR approach, although acknowledging uncertainty and change, does not provide direction for how it should be handled within the EcoQ and EcoQOs. Perhaps most importantly, the OSPAR approach *de facto* asks the scientific community to address political and social objectives, tasks which the ICES approach explicitly reserves for managers and their consultation mechanisms. WGECO expects that there will be problems with implementation of the OSPAR EcoQ and EcoQO framework in future as well, that may be amplified by these differences in approach to scientific advice. Although the OSPAR framework is developed as an overall framework for safeguarding the ecological health of marine ecosystems independently of the human activity threatening the system, many EcoQOs cannot be achieved without substantial cooperation by the fishing industry, and major changes in approaches to fisheries management (see Section 5.4.1). To facilitate such cooperation, the advisory approaches in support of the two frameworks should be as similar as possible. In that context, it is of particular concern that the Green Paper on the Common Fisheries Policy, although mentioning ecological quality as a concept, gives no role to EcoQs or EcoQOs in any stage of developing or implementing fisheries management policies and practices.

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6 SEABIRDS AND MARINE MAMMALS IN AN EcoQO-FRAMEWORK

At the 2000 meeting of the Planning Group for the Ecological Quality Objective Request (PGEQO), the Terms of Reference[2] assigned to WGECO with respect to EcoQO metrics for marine mammals and seabirds were revisited. It was concluded that WGECO should address this issue at a larger scale than the individual species levels that WGSE and WGMMPH would cover (ICES, 2001d). Thus the original WGECO Terms of Reference b) and c) were superseded. PGEQO considered that WGECO might not add much by trying to revisit issues for which WGSE and WGMMPH would have greater expertise. Rather WGECO would have the expertise to consider seabirds and marine mammals in a wider ecosystem perspective. WGECO decided to address the ToR by:

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[2] ToR b)

1. provide recommendations for appropriate EcoQO <u>indices</u> for marine mammals, and suggestions for appropriate EcoQOs for North Sea mammal populations;
2. prepare provisional estimates of current levels, reference points, and targets for the EcoQO indices identified in 1.

ToR c)

1. provide recommendations for appropriate EcoQO <u>indices</u> for North Sea seabird populations, and suggestions for appropriate EcoQOs for North Sea seabird populations;
2. prepare provisional estimates of current levels, reference points, and targets for the EcoQO indices identified in 1.

1) comparing the framework developed for EcoQs and EcoQOs (Section 5) with the approach taken by WGMMPH and WGSE;
2) commenting on the species metrics identified by WGSE and WGMMPH with regard to either their efficiency in detecting impacts or protecting the integrity of the community/ecosystem.

6.1 The Approaches taken by WGSE and WGMMPH

Drafts of both reports had been made available. The WGSE report was in a final state, whereas the WGMMPH report was only available in a very early stage of completion as their meeting was held simultaneously with that of WGECO. Neither report had been seen by the Advisory Committees. The approaches taken by both Working Groups are comparable (Table 6.1.1). In both cases the development of EcoQ starts with detecting general issues that are of concern for either seabirds or marine mammals.

Based on the OSPAR JAMP list, WGSE considered all possible classes of human activities that could affect seabird populations. This selection resulted in ten categories. For each category, potential EcoQ metrics, were considered. WGSE used nine criteria to screen potentially suitable EcoQ metrics. Their criteria match closely with the ones used by WGECO (see Section 5.3.2). For each of the final EcoQ metrics selected, WGSE tried to identify a reference level (often "pristine" levels), described the current status, and identified a target level for the EcoQ metric, if possible (Table 6.1.1). The target level chosen was that which WGSE considered achievable by current management, based on available evidence. WGECO observes that this differs from its interpretation of OSPAR target level (EcoQO) and the target levels proposed by WGSE could be regarded as "manageable levels" (which would form another category into the lower-most box of the EcoQO framework in Figure 5.1.1.1).

WGMMPH took a similar approach to that of WGSE, but focused more on metrics that described marine mammal populations rather than searching for metrics as descriptors of the state of the wider environment. After selecting and reviewing potential EcoQ metrics, six were selected for further development. WGMMPH tried to identify reference levels, their current status *sensu* OSPAR, and target levels for the selected EcoQ metrics. Just for one EcoQ, the population size of bottlenose dolphins in the NW North Sea, a target *sensu* OSPAR was defined.

Both Working Groups interpreted "reference level" in most cases as the pristine state or the state where human impact is minimal, but for a few EcoQs other reference levels were used. Reference levels were suggested for most EcoQs. There was insufficient information on cetaceans to allow estimates of total population numbers to be used for EcoQOs (CVs too high). Monitoring data on seabird populations seem to be sufficient. The target levels set for the EcoQOs differ in nature within the sets of both groups. In ICES terminology, Limit Reference Points are suggested for several marine mammal and one seabird EcoQO.

Both groups defined single species metrics at the population scale and applied them to as many species as possible. This approach increases the actual number of EcoQs to be further developed and ultimately used in management decisions (especially WGSE referred to quite a long list of bird species). However, these EcoQOs differ from the OSPAR framework in that both groups suggest that they should be used as triggers for further research on the causes of change, rather than as triggers for direct management action. In fact these EcoQOs do not really reflect management objectives or reference points but benchmarks for triggering further research.

Both groups recognised that EcoQOs have ultimately to be set by society through the political process. They respond to their terms of reference by interpreting the request to formulate provisional target levels by suggesting "manageable" levels or limit levels. The variety of levels that can be set on the EcoQ metric is potentially confusing; WGECO therefore advises that advice on EcoQs and levels needs to be carefully and precisely worded.

Limit Reference Points (LRP) may often be easier to develop from a scientific point of view. Some of them come directly from legislation and have a legal basis. Others may be developed from the dynamics of the populations concerned. The level of LRPs used by both groups are based on international standards (IUCN standards used by the mammal group, the widely used BirdLife International standards for the bird group). It may be wise to choose one approach rather than use two standards. If two standards are to be used, this choice would need to justified. Moreover, there is substantial debate within the marine science community regarding the appropriateness of the IUCN standards for marine species (see Section 7.4).

6.2 Evaluation of the Preliminary Results of WGSE and WGMMPH

From this overview, it is clear that all selected EcoQs refer to single-species metrics only (Table 6.1.1). This is an important observation because both groups did not rule out the possibility of developing community-based metrics.

WGECO considered this issue and could not suggest alternative community or ecosystem scale properties that would be of any greater help in the management of human activities in the marine environment with reference to marine mammals and seabird populations than those suggested for single species.

As for the WGMMPH report, many EcoQ values were not included in the draft available. WGECO was therefore unable to assess whether sufficient and reliable data are available to describe the current status of the EcoQs they proposed.

The match of some EcoQ metrics with the themes they covered raised some questions. The report of WGSE suggests the use of "breeding productivity of kittiwakes as an index for sandeel stocks in the North Sea". This would be a useful indicator within the foraging area of the kittiwakes, but not necessarily at the North Sea scale as the WGSE title suggests. Since the direct assessment of sandeel stocks is very difficult, it would not be straightforward to evaluate independently the accuracy or precision of seabird breeding productivity as an index for sandeel stocks at various spatial scales. Nevertheless, this EcoQ would be usable as a metric of availability of sandeels to predators, and recent decisions of fisheries managers in the EU are consistent with the information contained in this metric.

Table 6.1.1. Preliminary results of the Working Group on Seabird Ecology and the Working Group on Marine Mammal Population Dynamics and Habitats on the development of EcoQs and EcoQOs. Column headings are taken directly from both Working Group reports, although their use of the terminology may differ from the ones used within OSPAR or ICES (see Section 5.1.2).

Theme	Category	EcoQ/EcoQ metric	Current level	EcoQO Reference level	Target level
Pollution	Oil contaminants	Proportion of oiled guillemots among those found dead or dying on the beach	12–85 %	0 %	10 %
	Mercury	Mercury concentrations in eggs of selected seabird species	Various	no	no
		Mercury concentrations in body feathers of selected seabird species	Various	Possibly for situation in 1900	Suggested reference level
	Organochlorines	Organochlorine concentrations in seabird eggs	Various	zero	zero
Eutrophication					
Litter	Plastic particles	Number of plastic particles in gizzards of North Sea fulmars	Various, not well-known	0 %	10 particles within any fulmar of a sample of 40
Fisheries	By-catch				
	Harvesting food and predators	Index of breeding productivity of black-legged kittiwake as index for sandeel stocks	0.97	not known	LRP=0.5
	Increase in food supply				
	Mariculture				
	Habitats and ecosystem health	Seabird population trends as an index of seabird community health	Various	not known	LRP more than 20 % decrease within 20 years
Threatened and declining					
Hunting/harvesting					
Disturbance					
Introduced/conflicting species					
Climate change					
Community health	Harbour/grey seal	Population size	Increasing	0 % increase	More than 10 % decrease within 10 years
	Bottlenose dolphin	Population size in NW North Sea		Stable at a higher level than currently	>2 % increase per annum over at least 10 years
	Harbour/grey seal	Abandonment of breeding sites	Needs research	zero	Loss of more than 10 % of breeding sites within 10 years
	Harbour/grey seal	Number of births	Needs research	Current level	More than 10 % decrease within 10 years
	Harbour porpoise and other small cetaceans	No appropriate EcoQ selected	Needs research		
Contaminants	Seals	Concentrations of PCB, DDT, OC in body fat	Available	zero	Limit Reference Points are given
Fisheries	By-catch of harbour porpoise	Percentage of population killed (incidental by-catch)	Available	zero	<1.70 %
	By-catch of seals	Percentage of population killed (incidental by-catch)	Available	zero	<1 %

7 EcoQOs FOR FISH AND BENTHIC COMMUNITIES AND THREATENED AND DECLINING SPECIES

7.1 Introduction

The evaluation process undertaken in Section 5 provided a meaningful short-list of metrics that were considered the most appropriate descriptors of EcoQ of fish and benthic communities. Not only were they relevant to life history characteristics of species and their ecological functionality, but they also fulfilled a range of other criteria related to their implementability as EcoQOs. Table 7.1.1 provides the final list of metrics identified.

Table 7.1.1. Metrics selected by the process described in Section 5, scoring well on the seven selection criteria applied.

Fish communities
Length frequency (%age composition by size class; slope of size spectrum)
Mean length/weight of fish within specified limits
Presence of indicator/charismatic/sensitive species
Species abundance (k-dominance curves; species composition)
Maximum length (weighted mean L_{max} of community)
Mean and distribution of "body condition"
Benthos communities
Presence of indicator/charismatic species

The ten environmental issues identified by OSPAR as requiring EcoQOs are listed below:

1) Reference points for commercial fish species
2) Threatened and declining species
3) Sea mammals
4) Seabirds
5) Fish communities
6) Benthic communities
7) Plankton communities
8) Habitats
9) Nutrient budgets and production
10) Oxygen consumption

Issues 1 to 6, and 8 are considered to be directly influenced by fishing effects. Of these, other working groups within ICES are largely responsible for seabirds and marine mammals, and also for reference points for commercial fish. WGECO has included fish communities, benthic communities, and threatened and declining species in the concluding discussion of this topic as a broad interpretation of ToR e) but has not wanted to pre-judge the work that will be done over the next few months, particularly on the EcoQOs for habitats. The following sections review recent progress to provide further guidance on EcoQOs for these three environmental issues. Suggested metrics of EcoQ are then reviewed in the light of the framework suggested in Section 5. The final section describes a proposed approach to EcoQOs.

7.2 EcoQOs for North Sea Fish Communities

7.2.1 Introduction

Aspects of fish community structure may be described by a large number of metrics, and the effect of fishing activity on some of these metrics is currently under study. There is an extensive literature describing North Sea fish communities, and some large spatial and time-series databases. A pragmatic approach is to use these as the basis for evaluating fish communities, while recognising that this is only the most convenient method given the time available, and not necessarily the best. Initial discussion of this topic was greatly helped by a draft Netherlands working paper on potential EcoQOs for fish communities (Piet, 2001), which will be summarised in this section.

7.2.2 Summary of Piet (2001)

The study identified the following set of properties that covered both the structure and the functionality of the fish community (the list is similar to the one used by WGECO in Section 5):

Structure

- Biomass
- Size structure
- Species composition
- Species diversity
- Composition based on traits (e.g., life history, habitat preference, etc.)

Functionality

- Trophic structure
- Body well-being

For the development of EcoQ metrics, Piet (2001) used a pragmatic, largely data-driven approach. First, a suite of properties of ecological quality were established, and then the most suitable metrics were chosen based on a number of criteria:

- Representativity of a relevant aspect of the ecological quality.
- Quantifiability of the metric.
- Data availability through existing time series or historic data of a sufficiently large proportion of the area to which the metric is to apply.
- Causality (partitioning among effects of human activity, other forcing factors and inherent variability).
- Comprehensibility and communicativity (also to non-scientists, e.g., policy makers).
- Sensitivity for detecting gradual change (the signal should not be concealed by noise; possibility to determine a meaningful trend or variations in an objective manner.

Data from several North Sea surveys were available that differed in gears used and the area or time period covered. However, for calculating potential metrics, first quarter IBTS data were used (1974–1999).

7.2.2.1 Biomass

According to Piet (2001), the total biomass of the fish community present in the North Sea may depend on several factors such as the availability of food, water temperature or fishing effort. The total fish catch in weight per haul in a survey was suggested as a metric. The total catch per haul was calculated from the numbers caught per species and an appropriate length-weight relationship. This implies that the metric will reflect changes over time in species composition and size structure but not condition (Table 7.2.3.1). The total catch per haul for the whole North Sea showed considerable variation, with a relatively low biomass in the late 1970s/early 1980s followed by an increase towards a relatively high level in the 1990s. The time series of total biomass in different roundfish areas showed that within the North Sea there was considerable spatial variability.

According to Piet (2001), the functioning of a community with high biomass is not necessarily better than one with relatively low biomass, and likewise a pristine fish community does not necessarily have a higher or lower biomass than one that is impacted by human activities such as fisheries, eutrophication, etc. The high spatial and temporal variability provided no clue as to what the total biomass of a fish community in “optimal” or “pristine” state would be like. Hence, although there is no scientific basis for setting an EcoQO, they consider biomass an important metric of the functioning of the fish community and as such would certainly be worth monitoring, but would regard long-term trends with caution.

7.2.2.2 Size-structure

The study identified three metrics to describe the size-structure of the fish community:

- Slope of the biomass size spectra;
- Number or biomass in a specific size-class;
- Average size or weight.

All approaches to assess the change in size-structure of the North Sea fish community over time revealed the same pattern, which was a decline in abundance of large fish over time. This trend has been confirmed by studies that analysed change in the size distributions of roundfish and flatfish species using historic catch data (Rijnsdorp *et al.*, 1996; Rogers and Ellis, 2000), which found that the relative contribution of the larger fish has decreased since the early years of the 20th Century.

Considering that all metrics reveal the same trend, Piet *et al.* (2001) felt that the choice of the most appropriate metric for the size-structure of the fish community could be based on other criteria. Both the slope of the biomass size spectra and the average weight of an individual fish showed significant trends over time, with similar variation around the trend. They based the choice of a reference level on the time series available. Here the linear fit indicated a reference level of 230 grams average individual fish weight per individual in the early 1970s (Table 7.2.3.1). Although the average weight in a pristine environment should be higher, the authors felt that it was not possible based on available data and knowledge to come up with a reasonable estimate. Moreover, considering the measures necessary to at least reverse the current downward trend and realise a modest increase in average weight, it is hardly realistic to aim at this point for levels higher than this reference level.

7.2.2.3 Species diversity

In the study, three diversity indices were calculated per year for the North Sea fish community: Hill’s N0, N1 and N2. All indices showed an increase over time. Hill’s N0 showed a sudden step-wise increase in the late 1980s, and the other two indices showed a more gradual increase. The authors explain the difference as resulting from the increase in sampling effort in the late 1980s, because Hill’s N0 as a metric of the total number of species is highly dependent on sampling effort. Moreover, interpreting trends in species richness may be flawed because inconsistencies in reported species by different countries participating in the IBTS indicate that species identification has been unreliable (Daan, 2001).

Hill’s N1 and N2 are mainly dependent on the numbers of abundant and very abundant species, respectively. The explanation given for the increasing trend in these indices was that fishing mainly targets some of the most abundant species and that the additional mortality has resulted in an increased evenness and hence a higher index. Comparison with historic catches showed that in the early 1900s the fish community was slightly more diverse (Rijnsdorp *et al.*, 1996; Greenstreet and Hall, 1996), with Hill’s N1 being markedly higher in the past and Hill’s N2 within the range observed for present-day catches.

It was not always clear what changes in the fish community caused the observed changes in diversity, how this change affected the stability or productivity of the community, and to what extent it was induced by anthropogenic activities. Thus, the authors had difficulties in suggesting EcoQOs based on specific biodiversity indices.

7.2.2.4 Species composition based on traits

Piet (2001) considered the description of the fish community in terms of its biological traits an important ecological quality. The functional groups chosen were based on species characteristics pertaining to life history, habitat preference and biogeographic region. Habitat preference was captured in an index based on a distinction of two groups: (1) demersal (i.e., benthic and demersal), and (2) pelagic (i.e., pelagic, semipelagic, epipelagic, mesopelagic and bathypelagic). The index was calculated as biomass pelagic/biomass demersal. The index showed considerable variation over time, with high values at the start (late 1970s) and end of the sampling period (late 1990s) and lower values in between.

The biogeographic origin of a species was used to distinguish southern (i.e., Lusitanian, Atlantic tropical, Mauretanian) and northern (i.e., Arctic, Arctic/Boreal, Atlantic polar, Atlantic temperate, Boreal, Boreal/Arctic) species. The authors developed an index by dividing the biomass of “southern” species by the biomass of “northern” species. This index showed a trend towards a community with a high proportion of southern species. Although linear regression on all years showed that the trend was not significant ($p=0.16$), elimination of the 1991 outlier rendered a significant ($p=0.03$) trend.

These indices showed that the composition of the fish community in terms of functional groups changes over time and in some cases displayed trends. Piet (2001) concluded that the index based on the ratio of southerly/northerly species was the most sensitive to water temperature and represented a metric of the effect of water temperature on species composition of the fish community (Table 7.2.3.1). However, the contribution of human activities to the temperature changes and to the metric was not fully understood. Although biological traits are important ecological qualities of fish populations, the setting of EcoQOs, and the identification of reference levels, was not straightforward.

7.2.2.5 Trophic structure

Current theories of food-web structure and community regulation are based on a model in which species are described as homogeneous units, while the dynamic interactions among them form a network of consumer-resource relations. For most fish species, diets shift during early development and this complicates trophic interactions because a species may feed at different trophic levels during its ontogeny. Therefore information on both species and size is relevant when studying the trophic structure of the fish community.

Piet (2001) considered that the trophic structure of the fish community might be measured as the average trophic level of the fish community. Pauly *et al.* (1998) suggested that overfishing of stocks at a higher trophic level (i.e., piscivores) may result in refocusing of fishing effort on planktivores and lead to a corresponding decline in the average trophic level of the landings. “Fishing down the food chain” may significantly disrupt the food web and models suggest that it may have cascading implications for the stability of stocks and ecosystems (Christensen, 1996).

However, the authors conclude that quantification of the trophic level of each species- and size-specific trophic group is not straightforward. For many species the necessary information was lacking, or not even relevant given the shifts in trophic niche of many fish populations. Moreover, the determination of the trophic level of the fish community requires the intervention of some form of ecological model, representing hypotheses about the trophic interactions among species- and size-specific groups in the model. The question of how well the metric reflects the properties of the fish community cannot be dissociated from the question of how well the model represents the ecosystem (Rice, 2000).

7.2.3 Summary

The data in Table 7.2.3.1 summarise the properties and metrics that Piet (2001) considered relevant to EcoQs of the North Sea fish community. The table also presents reference levels for two of these metrics.

Table 7.2.3.1. Set of metrics to monitor the ecological quality of North Sea fish communities with current values and reference values. All values are based on the first quarter IBTS.

Metric	**Value of the metric**	
	Present	**Reference**
Average weight of individual fish (g)	60	230
Hill's N0	9.5	
Hill's N1	2.6	
Hill's N2	2.0	
Average maximum length (cm)	38	42
South/North ratio (x100)	2.5	
Pelagic/Demersal ratio (x100)	63	
Total biomass (kg/haul)	276	

The availability of survey data for the North Sea has obviously enabled a wide range of potential quality objectives to be evaluated. Most refer to community metrics involving the size structure of the populations and offer hope that meaningful objectives and reference levels can be reached. The way in which these ideas link to the WGECO framework is evaluated in Section 7.5.

7.3 EcoQOs for North Sea Benthic Communities

7.3.1 Introduction

Although much of the North Sea benthic environment is sedimentary in nature, a full consideration of EcoQOs for benthic communities must recognise that there are diverse habitats within this broad classification (see Section 3). Thus, a complete discussion of benthic EcoQOs in the North Sea must include habitats associated with the shallow and dynamic coastal waters, intertidal flats, offshore coarse environments and rocky reefs, and deep-water sediments such as those in the Norwegian Trough. Furthermore, a complete review of benthic communities must consider invertebrate infauna at a range of scales, as well as the macro-epibenthos and mega-epibenthos, which should include benthic and demersal fish populations. We provide here a summary of the study by de Boer *et al*. (2001), who provide a first attempt to develop EcoQOs for benthos, and thus serves as a useful starting point. Nonetheless, de Boer *et al.* (2001) develop another independent framework which must be modified further to correspond directly with the EcoQO framework.

7.3.2 Summary of de Boer *et al*. (2001) with comments

The study starts off with considering the benthos of the North Sea and the various schemes to classify the benthos. The authors settle on the scheme devised by Kunitzer *et al.* (1992), which identifies eight benthic community types. They acknowledge that this scheme ignores a large number of potentially important ecological habitats including kelp forests, sub-tidal rocky reefs, inshore and estuarine sediments, and intertidal areas. The classification also ignores epibenthos and is not consistent with either the MNCR Marine Biotope Classification (Connor *et al.*, 1997) or the EUNIS scheme (see Section 3).

The report then addressed which human pressures are most likely to influence benthic communities and proposed six measures that can be used to assess the status of benthic communities. Three measures (i to iii) address issues relating to species diversity and three (iv to vi) try to characterise community structure and function. The measures are: (i) species diversity as shown by Shannon-Weiner; (ii) abundance of fragile, vulnerable species; (iii) incidence of scar damage in *Arctica islandica* shells; (iv) the ratio of r- to K-strategists as shown by the W statistic derived from ABC curves; (v) abundance of opportunistic species; and (vi) the *Vas Deferens* Sequence Index in female *Nucella lapillus.* De Boer *et al*. (2001) noted that not all proposed indicator taxa (ii and v) will be present in all eight communities and suggested that EcoQOs and reference levels should be set differentially for each community type.

The report states that “EcoQO reference levels should represent the situation under minimal human impact” and advocates the use of values derived from the 1986 data series (Kunitzer *et al.*, 1992) as a basis for EcoQOs. In doing so, the authors ignore any of the other time series data sets that are available and that may provide information on a situation more close to the “minimal human impact”. If the assumption is that the eight community types are true reflections of real, distinct systems, then data from any station/study in one of the areas should be representative of the whole and therefore valuable in the consideration of reference levels. The other assumption made is that the 1986 values are lower limits for (i) and (ii) and higher limits for (v). It is by no means clear that human impacts have led to a decrease in diversity across large areas of the North Sea. It is widely recognised that natural disturbance is important for promoting diversity (Connell, 1978; Thrush, 1991; Hall *et al.*, 1994), through co-existence in a spatial mosaic. Anthropogenic disturbances can act in the same way, and a moderately impacted ecosystem can show either an increase or a decrease in diversity. The following metrics were considered:

- *Species Diversity as indexed by Shannon-Weiner*

De Boer *et al*. (2001) acknowledge the wide range of indices available that provide measures of “species diversity” and elect to use the Shannon-Weiner (H’) function primarily because of its wide usage in the literature and the fact that its limitations are well known. They recognised that measures of diversity are method/sample-effort dependent and will vary between community types. In spite of the well-described non-normality of this metric, the authors proceed to present means, and even standard errors, of H’. While acknowledging that human impacts can increase diversity, the presumption is made that high diversity indicates low levels of anthropogenic effects.

- *Abundance of fragile, vulnerable species*

This metric, and the subsequent one, focus on the physical impact of towed benthic fishing gears on the benthos. A number of short-term experimental studies (e.g., Lindeboom and de Groot, 1997) have shown dramatic levels of mortality on certain fragile taxa. The list proposed by the authors includes target species such as *Ostrea*, *Cancer*, *Hommarus*, and *Nephrops*. Many of the species listed are not recorded in the 1986 survey data, as they are epibenthic. This means that no reference levels can be set from this source.

- *Incidence of scar damage in* ARCTICA ISLANDICA *shells*

Arctica is a long-lived species that is recorded from a number of the community types. It has the capacity to recover from some physical impacts, but this leaves a scar on the shell. A relationship is available (Witbaard and Klein, 1997), which could allow this incidence to be used to quantify the level of impact occurring from heavy fishing gears.

- *Ratio of r- to K-strategists as indexed by the W statistic derived from ABC cu*rves

The underlying assumption of this metric is that an impacted community will have more r-strategists than an un-impacted one. The authors selected the W-statistic, derived from ABC analysis (Clarke and Warwick, 1994), as the most appropriate measure.

- *Abundance of opportunistic species*

This measure was suggested to provide information that complements the information from the ABC analysis. In particular, the authors focus on "small, opportunistic species". The need to develop different suites of metrics for each community is recognised, as is the need to then establish community-based reference levels. There is no discussion of the circularity inherent in selection of indicator taxa from the same data set that is then used to set their reference levels.

- VAS DEFERENS *Sequence index in female* NUCELLA LAPILLUS

This is the only metric proposed by de Boer *et al*. (2001) that addresses coastal and hard substratum communities. This measure is targeted at a single issue—of coatings containing TBT. It seems strange to target this issue using this index.

7.3.3 Summary

It is clear from this report that, in the short term, the most achievable measures of ecological quality are likely to be those which measure some aspect of species dominance in the community, and those which link the presence of fragile benthic species with direct impact of fishing gears. The way in which these ideas link to the WGECO framework is evaluated in Section 7.5.2.

7.4 EcoQOs for North Sea Threatened and Declining Species

7.4.1 Introduction

The availability of a working paper on threatened and declining species, and the recognition that fishing effects can impact individual fish and benthic species as well as communities, encouraged the WGECO to consider this topic. Threatened and declining marine mammals and seabirds were not considered.

To date there has been limited progress towards the development of ecological quality objectives for threatened and declining species in the North Sea (Lanters *et al*., 1999). Attempts to list and categorise threatened and declining species are relatively well developed, especially by the IUCN and the Bern Convention, OSPAR and a number of national "red listing" programmes. IUCN uses a system of standardised, quantitative risk criteria, while work in OSPAR has tested the subjective selection criteria produced at meetings in Texel (1997), Horta (1999), and at the 1999 OSPAR IMPACT meeting. Within the OSPAR region, the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) has listed several fish species for which trade is prohibited or restricted (World Conservation Monitoring Centre, 1993), although only two of these (the coelacanth *Latimeria chalumnae* and totoaba *Cynoscion macdonaldi*) are exclusively marine fish. The applicability of the criteria used by CITES when evaluating the risk status of marine fish and invertebrates is currently under review (FAO, 2000). The criticisms of the CITES criteria would apply equally to those of IUCN, from which they were derived (FAO, 2000).

Within Europe, the Corine Biotopes Project, used as the basis for species and habitat selection for the Natura 2000 network, listed 43 species in a checklist of threatened fish, but all species were either freshwater, estuarine or diadromous (i.e., fish that migrate between freshwater and the sea). No exclusively marine fish was listed. Within Europe, the rare fish of the Wadden Sea have been described and listed (Berg *et al.,* 1996), and within the British Isles, eight species have been identified as threatened or suspected of being threatened (Swaby and Potts, 1990). Recent developments in endangered species management by the American Fisheries Society suggest the use of the intrinsic rate of population increase to allocate fish species to a range of productivity categories (Musick, 1998, 1999). This helps to distinguish declining species with high rates of production, such as those of commercial importance, from others that are less able to recover stock size.

7.4.2 Summary of Gubbay (2001)

The working paper by Gubbay (2001) has taken further the development of EcoQOs for threatened and declining species, and this section describes progress made, and suggests a way forward. The approach taken was a pragmatic one, and used the output of a range of existing selection criteria as the basis for producing a short-list of threatened and declining species. It is logical to assume that strictly protected or endangered/vulnerable species previously identified by IUCN, the Bern and Bonn Conventions, OSPAR and other national programmes should provide a comprehensive species list. The list was further refined by excluding commercially exploited species and those which have a brackish water phase or which do not occur in the North Sea. A clear link between one of six human activities (OSPAR, 2000a) and the decline of a species was also used as a final criterion to provide a possible explanation of the reasons for decline. Based on these criteria, Gubbay (2001) identified fifteen species, which were considered to be particularly suitable for the preparation of EcoQOs. The list comprised macrobenthic invertebrate species as well as teleosts and elasmobranchs (Table 7.4.2.1).

Table 7.4.2.1. Threatened and declining species identified on international "red lists" that could be used to inform EcoQOs (Gubbay, 2001).

Lithothamnion corallinoides	maerl
Lithothamnion calcareum	maerl
Zostera marina	eelgrass
Alcyonium digitatum	dead man's fingers
Lophelia pertusa	deep-water coral
Atrina fragilis	fan mussel
Cetorhinus maximus	basking shark
Dasyatis pastinaca	stingray
Galeorhinus galeus	tope
Mustelus mustelus	smooth hound
Myliobatis aquila	eagle ray
Prionace glauca	blue shark
Raja batis	common skate
Somniosus microcephalus	Greenland shark
Trachinus draco	greater weever

The list included species that had already been identified for conservation and others that had not previously been identified as threatened or declining throughout the North Sea. Elasmobranchs are correctly identified as those known to be under the greatest threat, although the author acknowledges that some species, such as tope, skate and basking shark, have been, or still are, subject to commercial fisheries and should therefore be included in OSPAR issue 1 (reference points for commercial fish species). The greater weever has also been identified as a species that has undergone recent decline in the southern North Sea, and, although occurring elsewhere in the Northeast Atlantic, is not an abundant species. The reasons for decline are unclear, and there is no obvious management measure that could be applied in order to restore the populations of greater weever.

While several of the invertebrate species are entirely appropriate for potential EcoQOs, others are less useful. For example, dead man's fingers, a relatively robust colonial Cnidarian, have been listed as declining in the Dutch and German Wadden Sea but are very abundant in the Channel, Western Approaches, and the Central and Northern North Sea. This illustrates the difficulties of using local metrics of decline to infer species status on a larger scale. Similarly, the fan mussel is a species that has the northern limits of its distribution in the North Sea and is more abundant to the southwest of the region. As species at the outer limits of their normal distribution generally show greater variability in population size than nearer the centre, they may not be the best choice of species to use as an ecological objective.

7.4.3 Summary

In accordance with the conclusions of the Scheveningen Workshop, the general EcoQOs suggested by Gubbay (2001) for threatened and declining species are related to the Reference Level, which is defined as the species abundance when the anthropogenic influence on the system is minimal. While it is certainly important that objectives for this group of species highlight the need for improvement in population status, it is not clear that it is necessary or even achievable to restore populations to their unperturbed size and/or extent. These are societal decisions that cannot be made here. However, the author makes a basically sound proposal for an overarching EcoQO, viz. "an absence of threatened and declining species in the North Sea where the principal causes of threat or decline are linked to human activities".

7.5 Application of the WGECO Framework

Our approach to setting EcoQOs is described in Section 5. The following sections describe the application of this framework to fish communities, benthic communities, and threatened and declining species. It contains WGECO recommendations for the further development of EcoQOs for these three issues.

7.5.1 Fish communities

The ecological quality of fish communities can be described by a broad array of metrics, including the relative abundance of individuals, their species membership, the biological traits of individuals, and their life history strategy. The number of realistic metrics, however, is restricted by the information that is available for these communities from existing surveys, our understanding of the processes involved, and our ability to communicate complex metrics effectively (Table 7.1.1). Life history characteristics that involve fecundity, for example, are available for only a few species, and body condition and growth are not known for all species. When data for a specific variable are known for only a subgroup of species, it should be judged whether the subgroup is representative of the fish community.

The two metrics, average weight of individual fish and average maximum length, are those which meet the scientific standards that were set and thus were considered by WGECO to be the most suitable metrics of community structure. They describe key features of the relative abundance and size distributions (Section 5.5.3).

Fishing is probably the human activity that affects the fish community most. Fishing is size-selective because the gear targets the larger individuals and allows the smaller ones to escape. As a result, fishing tends to change the size structure of a community resulting in a decreased average body size. The size-specific mortality caused by fishing also affects the species composition through differences in life history parameters. This occurs in spite of the relatively unselective nature of the fisheries with regard to target species relative to other species with similar general morphology and habitat usage patterns.

Several of the aspects of the fish community represented by different metrics appeared to be related and could be traced to one specific type of human activity; fishery induces size-specific mortality which changes the size-structure of the population. Therefore, the proposed metrics for the North Sea fish community are the average weight of individual fish and the average maximum length. From a conservation perspective, appropriate EcoQOs would move these metrics towards a larger proportion of large fish and would improve fisheries yields. Neither metric would discriminate between treatments that simply allowed individuals of exploited species to grow larger (and live longer, i.e., lower mortality) and treatments that changed the species composition towards a higher proportion of species with larger maximum possible weights and lengths (redistributing mortality across species, away from ones with greater maximum sizes).

7.5.2 Benthic communities

7.5.2.1 Introduction

In considering the broad aims of ecosystem management with reference to the benthos, the most important community metrics appear to be the species composition (including the presence of fragile, opportunistic, and keystone species), and its productivity and trophic structure (Section 5.3). It must also be emphasised that few, if any, studies provide a holistic picture of the benthic community. This is largely the result of the constraints of the sampling regimes required. Meio-infauna, macro-infauna, and epibenthos of soft sediments are rarely recorded in the same surveys, let alone in a way that would allow synthesis of the data into a “community picture”. The situation is even more problematic when one considers hard grounds not amenable to grab/core sampling. There are two possible approaches to setting EcoQOs under these conditions. One alternative would be to focus on one aspect of the benthic community, and assume that if this component meets the EcoQ then other parts of the community will also conform. Alternatively, one could set EcoQOs for each component of an area: meiofauna, infauna and epifauna of sediments, sessile epibiota and mobile epifauna for rocky areas. The latter approach would greatly increase the number of EcoQOs required and might involve problems of consistency among components in their response to management measures.

7.5.2.2 Metrics of EcoQ

In Section 5.3 only one metric, the presence of indicator or sensitive species, was identified as a good metric of ecological quality in benthic communities (Table 7.1.1). There are several indicator species, often consisting of structural biota such as corals (Fosså *et al*., 2000) and epifaunal organisms that are known to be sensitive to bottom fishing disturbance (Freese *et al*., 1999). These species are often apparent in bottom photographs and videos (Collie *et al*., 2000). The use of indicator species obviates the need to identify all species in benthic samples. However, in some benthic communities, there may be no obvious indicator species, suggesting that this EcoQ may not be comprehensive. Also, some epifaunal species that may make good indicators may have been removed by past fishing practices, yet present fishing practices may continue to impact benthic ecosystem function.

7.5.2.3 Metrics that might be developed further

The presence of indicator or sensitive species cannot measure all the properties of benthic communities. Three other metrics measuring different properties of benthic communities scored quite highly using the framework in Section 5.3. These metrics were biomass, K-dominance curves, and the presence of non-indigenous species. Adoption of these as metrics of benthic EcoQ may address some of the shortcomings of the application of “the presence of indicator or sensitive taxa”.

Biomass per m^2 is an aggregate measure of the benthic community that does not necessarily require all species to be identified. Biomass is also a component of benthic productivity. It is, however, difficult to measure benthic productivity directly, as often the benthic production is estimated by multiplying P/B ratios by biomass measurements. Hence, biomass is a more direct measure of benthic ecosystem quality. For example, significant decreases in benthic biomass have been measured in response to bottom fishing (Collie *et al*., 1997) and to sediment extraction in the eastern English Channel (Desprez, 2000). Disadvantages of using biomass as a metric are that environmental and anthropogenic impacts on biomass variations may be confounded, and time series of benthic biomass are also not available in most locations.

K-dominance curves may provide a useful measure of changes in species diversity in benthic communities. As this index was derived to measure impacts on benthic communities, it is clearly applicable as a measure of ecological quality of some parts of the benthic community. K-dominance curves are obtained by plotting cumulative ranked abundance against the log of species rank (Lambshead *et al*., 1983) and the shape is a direct function of species relative abundance. Perturbations allow a subset of tolerant species to persist while the intolerant species disappear or become rare, hence the curve is expected to change in a predictable direction in response to disturbance. Shifts in K-dominance curves have been demonstrated in response to pollution (Warwick, 1986) and to experimental beam-trawling disturbance (Kaiser and Spencer, 1996). A potential disadvantage is that this graphical representation is somewhat difficult to comprehend and to communicate to policymakers and other non-specialists.

The presence and abundance of non-indigenous species may also be a useful metric of ecological quality. Non-indigenous species, both invertebrate and fish, have been widely spread by the discharge of ships’ ballast water (ICES, 2000) and in some areas have markedly altered benthic food chains and community structure (ICES, 2000). As an example, the slipper limpet *Crepidula fornicata* has largely replaced the native oyster *Ostrea edulis* in Poole harbour (southern England) (ICES, 2000). The spread of non-indigenous species is clearly caused by human activity, but it can be very difficult to manage this activity and the invasion of indigenous species is likely to be impossible to reverse.

This suite of four metrics provides potentially useful measures of ecological quality in parts of the benthic community, but their practical application is limited by the history and intensity of benthic sampling across all parts of the meiofauna and macrofauna. While these metrics are most applicable to the benthic macrofauna and epibenthos, in principle they could also be applied to the meiobenthos, but there has been much less sampling to support their use in this part of the benthic community.

By not meeting our criteria fully, the inclusion of the additional three metrics is a weaker basis for ensuring that real conservation results from the management actions guided by advice developed within this approach. This does not mean that it is necessarily a bad approach, but WGECO expects it to be an approach with higher risk than would have occurred had more metrics met the required criteria.

7.5.2.4 Adding spatial dimensions

The purpose of EcoQOs is to assist in the development of an ecosystem approach to management (see Section 5). OSPAR has pragmatically adopted ten issues for which it believes that, if addressed by well-selected EcoQOs, ecosystem management will be assured. The problem is that this review has found that we are far from being able to identify scientifically sound and reliable metrics for enough of the benthic species and communities to address the ten issues effectively. It may, however, be possible to assure the quality of the ecosystem without necessarily having EcoQOs for all ten themes. Operationally, a restricted number of specific EcoQOs will ensure a more streamlined procedure. With regard to the benthos, we may therefore say that although it is critical that ecological qualities such as biomass, K-dominance curves, the presence of indicator or sensitive species, and the absence of non-indigenous species, are maintained, there is a more pragmatic approach that can be applied until science is able to support a full EcoQ and EcoQO framework.

Analyses of all the principal North Sea benthic data series which cover infauna and epibenthos (Buchanan, 1963; Basford *et al.*, 1990; Kingston, 1992; Kroncke and Rachor, 1992; Kunitzer *et al.*, 1992) highlight the critical role of the sediment in determining benthic community distribution. This is obviously superimposed on the broad framework of biogeographical and depth gradients (Pearson and Mannvik, 1998). The dynamics of these assemblages are also highly influenced by the quality of the overlying water, both in terms of pollution stress and also in response to variations in climate and pelagic dynamics (bentho-pelagic coupling) (Pearson and Rosenberg, 1986). A logical extension of this line of reasoning is that, if processes in the water column are managed and measures are taken to ensure that the benthic environment is not significantly altered (i.e., by sediment deposition, trawling impacts), then benthic ecological quality will be assured. We feel that to this should be added a clause recognising the need for a suitable larval supply and the fact that this supply may be spatially distant from the area under consideration. It must be emphasised that in this approach the benthic environment (habitat) comprises not just the physical features but also the biotic assemblage.

Adoption of this approach has a number of advantages:

- It reduces the total number of EcoQOs that need to be managed while still having a reasonable (if unquantified) likelihood of preserving benthic ecological quality;
- It is operationally easier to manage water quality and habitat destruction than directly managing the processes behind the more complex graphical presentations such as benthic K-dominance curves;
- It has an explicit spatial dimension and so is amenable to the spatial application of management techniques.

This approach is based on the relationship between benthic communities and their environment. It reaffirms that threats to benthic ecological qualities must be managed, but avoids the setting of EcoQOs based on any specific metric of the benthic community. The approach has obvious parallels with terrestrial conservation biology, where protection of a species' habitat is considered essential to protecting a species' population. The parallels between seafloor obligates and their structural habitat with terrestrial species is not a coincidence; "habitat" for both types of organisms has a potential stability over space and time that can be protected, in a way that "habitat" in the water column often lacks.

This approach does not obviate the need to monitor the properties of benthic systems which might provide an early warning of ecosystem disruption. High quality habitats may be a necessary, but may not be a sufficient, condition for stable and mature benthic communities. Such an approach also reinforces the need for effective Habitat and Water Quality EcoQOs. Habitat is used here in the sense defined in Section 3 and includes consideration of biotic associations (EUNIS levels 4–5). These EcoQOs must be developed and subjected to the rigorous selection process described here (Section 5). This approach does not lessen the need for effective management measures to protect threatened and declining species and to continue to monitor benthic communities. This approach may be seen as being pragmatic given our current level of knowledge about North Sea benthic communities but may also be seen as precautionary, as it will ensure protection of suitable habitats, provided that habitat EcoQOs are successfully identified and management measures are successful in protecting them.

7.5.2.5 Conclusion

There are at least three approaches to the setting of EcoQOs for benthic communities:

a) the use of a single metric, presence of indicator or sensitive species (Section 7.5.2.2);

b) the use of four metrics which provide a wider coverage of ecological qualities of benthic communities, but three of which fail to meet the strict criteria set by WGECO (Section 7.5.2.3); and

c) the use of EcoQOs for benthic habitats and water quality to provide the necessary protection to the ecological quality of the benthos (Section 7.5.2.4).

The first approach is more rigorous but very specific and suffers a number of limitations (Section 7.5.2.2). The second lacks the full rigour of the WGECO approach and so will provide a weaker basis for ensuring that conservation objectives are met. However, it covers a wider range of benthic community ecological qualities. The third approach avoids the need to identify specific measures of the community, and instead adopts a threats-based approach, recommending that protection of benthic community qualities is provided, at least initially, through the setting of rigorously developed and comprehensive habitat and water quality EcoQOs. These approaches are not incompatible, however, and WGECO favours the use of both the four metrics and the threat-based approach while further development of specific benthic EcoQOs is progressed, and the efficiency of each approach is tested in practice.

7.5.3 Threatened and declining species

Important ecological qualities of threatened and declining species are a measure of the population size, its trend and rate of change, and the assurance that current populations are sustainable. Invariably this requires information on population abundance collected either from catch statistics or independent surveys, with a sufficiently long time frame to assess the rate of change. The most important management objective for species that are under threat or are declining is to prevent further decline, and then restore population size and spatial extent. The reference levels for these metrics need to be debated but there are few alternatives to gradual improvement in population status.

Choosing the correct criteria for selecting species that are under threat is a crucial decision, and further elaboration of these criteria is required to select the most appropriate list of species for protection in the North Sea. Inappropriate criteria which do not take account of different species life histories can result in the selection of an incomplete list, ignoring some genuinely threatened species while including others in less need of action (Musick, 1998, 1999; IUCN, 1994; FAO, 2000). The list of species identified by Gubbay (2001), while identifying the elasmobranchs as requiring protection, also specify other species which are less appropriate, or less in need of protection. This largely resulted from the use of criteria that depended to some extent on the selection processes undertaken for different purposes by other bodies.

While recognising that there is a need to improve the population status of threatened and declining species, it is not at all clear to what level the populations should be restored. It is unlikely, however, that the reference level in an unperturbed environment will be a realistic or necessary objective. WGECO considers that there are insufficient data available for populations of threatened or declining species to identify levels that will ensure their long-term security, and inadequate guidance on what is the best reference level. The field of population viability analysis has been developed to address exactly these questions for tetrapod species. However, the analysis methods are highly sensitive to non-linearities and density dependence in population dynamics equations (Bergman *et al.,* 1992), and such attributes are routine in fish population dynamics models, such as stock-recruit equations. There is no doubt, however, that the EcoQ of these populations, i.e., their absolute population size, trend and rate of change, must be recorded in detail.

Gubbay (2001) suggested an overarching EcoQO based upon the “absence of threatened and declining species in the North Sea where the principal causes of threat and decline are linked to human activities”. This would seem a useful suggestion, because the single metric might be the number of such species and the objective to reduce the number to zero. WGECO did not discuss the potential at any depth, largely because threatened and declining species did not stand out in the terms of reference. However, this possibility should be evaluated further in the future, because it would overcome the problem of setting EcoQOs (and consequently collecting detailed data) for each individual species identified as belonging to this category. Only appropriate criteria for inclusion or exclusion need be developed. This will not necessarily be scientifically straightforward, nor without debate (FAO, 2000).

7.5.4 Concluding thoughts and the way forward

EcoQOs are a tool for integrating the management of human activities at the ecosystem level. The OSPAR definition of EcoQOs makes it clear that objectives need to be seen as components of an integrated set, which together comprise the overall expression of the structure and function of the marine ecosystem. Due to the complexity of the ecosystem, a pragmatic approach to setting EcoQOs is to target specific, separate issues as components of the ecosystem (OSPAR, 2000b). Most importantly, the metrics must be related to specific human activities, so that plausible and effective management solutions are advanced and the link between the human activity and the impact is clearly understood.

The preceding sections describe our recommendations on the development of EcoQOs for fish and benthic communities, as described in TOR e). We have also included in this section an analysis of the issue “threatened and declining species” for two reasons. First, we considered that many of the potential candidates in this category were either fish or benthos, and as such may anyway be discussed in separate sections, and secondly, a consultant’s report on threatened and declining species was made available to WGECO for consideration.

Throughout this report, we have aimed to provide a limited number of metrics that can be made operational now. WGECO does not feel that it is necessary to select several EcoQs for all components of the system, especially where some characteristics of a particular metric, i.e., its degree of association with a human impact, and the complexity in deriving it, will largely prevent its use. There are also clear disadvantages in trying to apply some of the metrics that are described in Sections 7.2, 7.3, and 7.4, as EcoQOs. For example, if the metric is a good descriptor of some aspect of the community but its link with one or several human impacts is weak and poorly understood, there is little chance that the EcoQO will be effective. This failure will allow the approach to be undermined by those groups with an interest in maintaining the *status quo*.

We have identified several other metrics which may be useful in the medium term, but which we do not feel are sufficiently developed for immediate use. Given the time-scale for the current process of developing EcoQ, WGECO feels 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 (see Section 5.3.6). In the long term, there is a need for the development of new metrics to describe these key ecosystem qualities.

There is another reason why it is not thought necessary to set multiple objectives for all parts of the marine ecosystem. There are key environmental qualities such as habitat integrity and water quality which underpin the structuring of the marine ecosystem, particularly its benthic components, and influence communities at all trophic levels. By targeting these key features, and setting specific objectives to ensure that there is maintenance and improvement in quality, we can do a great deal to ensure the natural composition and function of the community.

The group did not have as a TOR the identification of EcoQ for the issue of marine habitat. However, we await developments on this topic with interest because human impacts have an implicit spatial component, and the maintenance of spatial integrity of marine fauna is identified as an issue in the Convention on Biological Diversity. In our discussions of habitat classification and integrated environmental management (Section 3), WGECO has illustrated the importance of using GIS presentation of impacts as powerful tools in the management of habitat quality and thereby evaluating the success of specific EcoQO.

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8 FOOD FOR THOUGHT

8.1 Proposal on Screening Metrics

8.1.1 Introduction

A series of trials for metrics that are to be considered for EcoQ status is proposed. These are not meant to replace or contradict the seven (or eight) criteria listed in Sections 5.2 and 5.3, but rather serve as an operational structure further testing and selection.

The comparisons are set up as a cascading series of tests. A trial metric, after passing a series of increasing restrictive tests, would be given provisional EcoQ status, as an informative metric. This would indicate that it is useful and informative. EcoQs would be further tested to ascertain if they would be suitable for management and the formulation of accompanying EcoQOs. We call this full EcoQ status. It would require the ability to define limits or desirable states and the ability to be manipulated. This sequence brings together ten days of discussions and accumulating experience during the meeting, but is proposed without having been tested and may well be amended with experience. The order is suggested as being increasingly restrictive, analogous to a series of sieves. For the duration of this section we will use EcoQ(I) to denote one which is informative but unlikely to be useful in a management context and EcoQ(F) for a full EcoQ.

Table 8.1.1.1. Cascading tests for indices; if a test is passed go down to the next.

	Status	Test	Comment
	Metric		
↓		Preliminary screening	expert scientific opinion
↓		Detection screen	does the index detect a known event
↓		Signal quality screen	stability, error rates, S/N…
↓		Uniqueness/contribution	How does it act in concert with others
↓	**EcoQ(I)**		
↓		Manageability	Can objectives be defined?
↓		Interference/contribution	How does it act in concert with others
	EcoQ(F)		

Most of the following comments are based on case studies and real data as opposed to simulations. When a specific topic is amenable to simulation, it will be so indicated.

A simulation would have to be fairly sophisticated to serve as a basis for index comparisons. Top-down and bottom-up controls, prey switching, susceptibility to exogenous factors (temperature, pollution…) and perhaps spatial structure might all be essential components of a simulation, if the simulation is to be informative about the performance of the metric. However, real-world experience tends to lack replication and controls, so simulations, although complex, will often be essential in full evaluation of candidate metrics.

Preliminary screening of metrics

Metrics may be classified as single species sole (sss), single species indicative (ssi), emergent, or aggregate. The single species (sss) indicator is only applicable at the species level and is not asserted to have significant ecological implications – for example, the abundance of a non-rare, not highly coupled, non-charismatic species. The indicator single species is of broader interest. The emergent indices are only applicable at the community or higher level; an example is r/K ratio. The aggregate metrics are aggregates of single-species metrics (mean length across species)

Detection screen

Case studies, particularly for the North Sea, require an agreement that an event has happened for which management action was justified (whether action was taken or not). Events in this context are things like (excess) fishing and pollution events. The event must be expressed at the community level to test “emergent” property candidates. Does the reduction (say to 10 % of historical levels) of a single keystone predator signal an important change to the ecosystem? Once there is agreement on what “events” should have been picked up by useful metrics, which metrics showed trends or changes in value that corresponded to the “events”? The first screening includes those indices that experts feel have ecological relevance and are likely to perform well. These can be compared (statistically) to the timing of the onset of the event. If time series are not available, spatial data may be used for comparison: areas where the event took place versus areas where it did not.

Signal quality screen

This screening consists of estimation of signal/noise sensitivity, false positive rates (hysteria), false negatives (misses). Replicate trials and controls are difficult to find in ecosystems. Areas and seasons tend to be fished for decades. Closed areas and fishery closures offer some controls (in the experimental sense) that could possibly be used to assess the responsiveness of the signal to a perturbation. Signal-to-noise ratios need to consider a number of aspects of the noise. The noise may be estimated from bootstrapping or parametric analyses. The mean level of noise (or some index of a central tendency like the standard deviation) can be compared to the dynamic response of the metric, a traditional signal-to-noise ratio. This ratio gives an insight into the risk of the metric to miss real events. Also of importance are the tails of the noise distribution—what could happen on an unusually bad year. Is the noise correlated—will a number of bad years happen in a row? Insights are gained from meta-analysis of noise in related systems in an attempt to anticipate events that have not been seen in the analysed system. The more extreme events, especially correlated ones, may trigger false positives.

Uniqueness/contribution screen

If they pass the detection of event test, the next series of questions needs to be considered. Are the metrics independent? From one perspective managers do not need six indices changing at the same time, so one should pick the easiest to measure, or the most charismatic. On the other hand, if they are not perfectly redundant, it could be argued that a suite of related metrics might be more informative than any single one, just as meta-analysis is thought to add strength to analyses of numerous, individually noisy, data sets. However, whenever there are multiple indices, it is necessary to ask if they are contradictory, and if so, how to use models for resolving them or methods of integration, perhaps Boolean operators or weighted means, or fuzzy logic. Some metrics are model-based derivatives of others, for example, production may be composed of survivorship, growth and reproduction metrics. They will be correlated with others but the linkage is explicit. The signal detection attributes, or cost, could be used to discriminate among correlated metrics.

Integration of metrics will require some structure. The object would be to improve the reliability of detection with a combination of metrics. One model could be Boolean, any alarm will be carried through. This means that the most sensitive metrics will dominate. Another could be some sort of averaging weighted by importance or reliability. Fuzzy logic can also be of use in integrations of this sort.

Manageability screen

The scientific contribution to screening EcoQ(I)s as potential EcoQ(F)s may not be as important as social, economic and political considerations. One role for scientists is in the expected ecological effectiveness of a measure. How fast would the action prompted by the metric have an effect and how tightly coupled is the EcoQ to the control (fishing effort reduction, cessation of oil exploration). Ecosystems can behave in unexpected ways. Irrespective of effort reduction, populations may be in a new (stable) regime; for example pink salmon in Peterman (197x) or the failure of some Northwest Atlantic cod stocks to recover after eight years of closure.

A related issue in elevating a EcoQ to full EcoQ(F) status is the requirement of the ability to define relevant management objectives. As discussed with metrics of species diversity, it is very often hard to know what measure(s) is the right one to "correct" an anomaly in a diversity index, harder still to convince managers that the measure is essential, and perhaps hardest of all to convince stakeholders restricted by the measure to comply with it, if the debate centers on the value of a diversity index.

Interference/contribution screen

How potential EcoQ(F)s act in concert would be amenable to simulation. A simulation would have to be fairly sophisticated to serve as a basis for EcoQ(F) with applied EcoQO comparisons. Top-down and bottom-up controls, prey switching, susceptibility to exogenous factors (temperature, pollution, etc.) would be required and if possible spatial heterogeneity.

8.1.2 Concluding remarks

These proposals are suggested as a complement to the work carried out in Section 5. The steps of the screening do not match the properties used in Section 5.3 closely. Most of these properties (for example, "Comprehensive and communicable", or "Easily and accurately measured") are dealt with in the first step of the screen. On the other hand, several of the steps in the screen are collapsed in the Section 5.3 property "High response to signal from human activity...". Two of the remaining screens are focused on how groups of metrics work in concert, a characteristic not explicitly addressed in Section 5.3, but reserved for "later" consideration.

8.2 Evaluation of Potential for CPUE as a Management Tool

Several management measures that aim at restricting fishing effort are available to protect the ecosystem from the effects of fishing activities:

- Reduction of the fleet;
- Reduction of days at sea;
- Closed areas;
- TAC measures.

A suggestion was put forward to evaluate the potential of an alternative management measure using Catch Per Unit of Effort (CPUE) that aims at assuring a maximum yield while minimising the ecosystem effects of fishing. The idea behind this approach is that CPUE of a species can be considered a metric of stock abundance. Having established the relationship (while taking account of its non-linearity and ongoing technical improvements in respect of catchability), CPUE of the fishing fleet may be used to determine when stock abundance is near a critical level and additional measures such as a partial closure of the fishery need be taken. Such a system may create a strong incentive for fishermen to improve fishing efficiency as well as to land all their marketable catch, thereby decreasing the amount of “highgrading”.

To determine CPUE, catch and effort need to be quantified. Catches by species can be estimated from landing statistics. Effort can be determined by either (1) the days-at-sea available from logbook information or (2) data on fishing activities collected using Vessel Monitoring through Satellite (VMS). An advantage of the latter method is that effort can be more accurately recorded and it allows the skipper to decide not to fish certain areas while at sea in order to lower his fishing effort.

Studies of the microdistribution of fishing activities have shown that fishing intensity is extremely patchy, with relatively small areas where most of the effort is concentrated and large areas that are hardly fished. Studies of the effect of fishing activities on mortalities of non-target epibenthic species showed that the largest increase in mortality is realised by minor increases in fishing intensity in areas with low fishing intensity (i.e., the increase in mortality of benthic species from untrawled to trawled once per year or from once to twice per year is considerably larger than from 10 to 20 times per year).

The assumption is that intensively trawled areas exist because CPUE is relatively high in these areas. The locations of heavily fished areas have remained constant over time. The phenomenon of hyperaggregation of fish will, on the one hand, assure that in spite of ongoing fishing activities CPUE can remain relatively high in these areas; on the other hand, it should be realised that this may cause CPUE to remain high while stocks are declining.

Forcing fishermen to maximise their CPUE will be an incentive to avoid areas with low-catch rates and to concentrate in relatively small, intensively fished areas, where catch rates are high, thereby reducing the impact on the ecosystem. Hence, the proposed management tool may have several positive effects:

- the impact on (benthic) non-target species can be reduced by directing fishing effort away from the lesser trawled areas;
- improved reliability of landings-statistics will enhance the accuracy of fish stock assessment;
- improved dialog between science and the fishing industry.

Data are available to test some of the underlying hypotheses, which is obviously necessary before potential problems can be addressed.

9 THE FUTURE OF WGECO

9.1 The Evolution of WGECO over the Past Ten Years

For the past decade, WGECO has provided the ICES community with analysis and scientific support for advice developed from a broad constituency of experts. Our analyses have led to new advances in understanding of the ecosystem effects of fishing, while the support for advice has been highly regarded both within ICES and beyond. However, over time the nature and quantity of the work has changed. At recent meetings we have had less opportunity for collating data and analysing impacts and more time devoted to providing scientific support for advice to underpin policy developments at the ecosystem level and to advising on the strategic development of the science and policy framework. The need for such scientific advice (and the work to provide its technical foundations) is likely to continue and even accelerate in the near future. It therefore seems timely to review the working methods of WGECO. In doing so it is important to protect its key strengths—the dynamics and diverse membership—but also look at ways of meeting the new needs more efficiently.

The workload at recent meetings has hampered progress on strategic thinking and developments (see ICES, 1998, 2000a) as support for work driven by advisory needs has taken priority. Given the short time frames often associated with deadlines for such advice, it seems appropriate to increase the frequency of working group meetings which support the advice to annually. This will allow advice to be progressed in a timelier manner. It will also see a reduction in the workload of each meeting and so should allow shorter, but more focused meetings which have greater scope for development of strategic thinking and targeted development of science.

The framework within which WGECO operates is an integrative one, and it is increasingly difficult to address single factors such as fisheries impacts at the ecosystem level without considering other factors. It has always been the case that WGECO has drawn upon the diverse experience of its members to underpin the advisory role of ICES on a wider range of environmental issues. With international developments leading to more integrative, ecosystem-level, management frameworks, we feel this diverse expertise should be recognised by a formal broadening of the terms of reference given to the group, to allow the meetings to support advice which is integrative and ecosystem-based. The broader mandate would also help prevent the disciplinary diversity of WGECO, one of its greatest strengths, from being diluted as members are required by their home institutions to cover off different Working Groups with narrower mandates.

Recommendations:

- If the work of the WGECO is to become more responsive in providing timely science analysis to underpin policy advice, then consideration should be given to the adoption of annual meetings, the duration of which reflects the work to be undertaken.
- WGECO should continue to receive Terms of Reference which reflect its proven abilities (i) to integrate the analysis of human impacts on marine ecosystems into scientific support for an advisory framework, and (ii) to provide analysis of scientific issues which generates new levels of knowledge and understanding.

9.2 Proposed Terms of Reference for Future Meetings

a) Informed by progress being made within the OSPAR Biodiversity Committee and during the 5th North Sea Conference in Norway in early spring 2002, continue the work started at the WGECO meeting in 2001 to develop the scientific components needed for provision of scientific advice required by an EcoQO framework.

Rationale:

Broader political support for an ecosystem approach in the management of marine systems is to be expected in the near future. The development of an EcoQO framework for the North Sea by OSPAR may result in important implications for fisheries management and for the way fisheries advice is given. Because the ecosystem approach refers to the integration of all human activities, broad expertise is required to develop such a framework. WGECO felt that addressing the issue in 2001 is challenging, valuable and fruitful, and that it is capable of dealing with this issue.

b) To the extent possible, 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.

Rationale:

WGECO has addressed this problem in past meetings, most recently in 1994. In the ensuing years a great deal of new information and analyses have accumulated, but have not been consolidated by thorough review and integrative analyses. The QSR 2000 has highlighted the role of fisheries as the human activity with the single greatest impact on the North Sea ecosystem. It seems particularly timely for ICES to update its evaluation of the actual impacts of fishing, relative to other activities, rather than rely on an evaluation rapidly becoming outdated.

c) Continue the workplan to test hypotheses about which components of the marine ecosystem are most sensitive to bottom fishing impacts.

Rationale:

At its 2001 meeting, WGECO made substantial progress in developing a suite of testable hypotheses (Section 4.2). Having compiled this list, the logical next step is to test as many of these hypotheses as possible with data from different areas. Most of the available data are for fish communities, but these hypotheses should also be tested for benthic communities to the extent that benthic data are available. At the 2001 meeting, progress on this ToR was limited by the time demands of the other ToRs and by delays in obtaining the necessary data. WG members are strongly encouraged to collate data sets on fish and benthic communities, and fishing effort in the corresponding areas. These data should be brought to the meeting ready for analysis such that significant progress can be achieved in testing the hypotheses listed in Section 4.2.

d) In most fisheries, large amounts of fish and benthos are discarded. The amount of by-catch discarded relative to the catch varies greatly between types of fishing gears and what species the fisheries is targeting. Three meetings ago WGECO dealt with a Term of Reference regarding evaluating the impacts of discards and by-catch from fisheries in the North Sea. At that time few data on by-catch and discards were available, and few informative analyses could be undertaken. The problem of assessing the scale of by-catch remains important, however.

Rationale:

The Study Group on By-catch and Discard Information (SGDBI) was formed to estimate how much of the catch is discarded in various fisheries (e.g., ICES, 2000b). WGECO remains very interested in the magnitude of discarding as a fishing effect on the ecosystem, and on the consequences of discards and by-catch from fisheries on marine communities. When by-catch and discard information has been collated and quality-checked by SGDBI, WGECO would like to have the data made available to it for analysis and interpretation in an ecosystem context. At that time WGECO would like to have a Term of Reference to explore those data, and to suggest ways that they could be informative for ecosystem management.

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ANNEX 2: CONVENTION FOR THE PROTECTION OF THE MAINE ENVIRONMENT OF THE NORTH-EAST ATLANTIC, ANNEX V ON THE PROTECTION AND CONSERVATION OF THE ECOSYSTEMS AND BIOLOGICAL DIVERSITY OF THE MARITIME AREA

ARTICLE 1

For the purposes of this Annex and of Appendix 3 the definitions of “biological diversity”, “ecosystem” and “habitat” are those contained in the Convention on Biological Diversity of 5 June 1992.

ARTICLE 2

In fulfilling their obligation under the Convention to take, individually and jointly, the necessary measures to protect the maritime area against the adverse effects of human activities so as to safeguard human health and to conserve marine ecosystems and, when practicable, restore marine areas which have been adversely affected, as well as their obligation under the Convention on Biological Diversity of 5 June 1992 to develop strategies, plans or programmes for the conservation and sustainable use of biological diversity, Contracting Parties shall:

a) take the necessary measures to protect and conserve the ecosystems and the biological diversity of the maritime area, and to restore, where practicable, marine areas which have been adversely affected; and
b) cooperate in adopting programmes and measures for those purposes for the control of the human activities identified by the application of the criteria in Appendix 3.

ARTICLE 3

1. For the purposes of this Annex, it shall *inter alia* be the duty of the Commission:

- to draw up programmes and measures for the control of the human activities identified by the application of the criteria in Appendix 3;
- in doing so:

    i) to collect and review information on such activities and their effects on ecosystems and biological diversity;
    ii) to develop means, consistent with international law, for instituting protective, conservation, restorative or precautionary measures related to specific areas or sites or related to particular species or habitats;
    iii) subject to Article 4 of this Annex, to consider aspects of national strategies and guidelines on the sustainable use of components of biological diversity of the maritime area as they affect the various regions and sub-regions of that area;
    iv) subject to Article 4 of this Annex, to aim for the application of an integrated ecosystem approach.

- also in doing so, to take account of programmes and measures adopted by Contracting Parties for the protection and conservation of ecosystems within waters under their sovereignty or jurisdiction.

2. In the adoption of such programmes and measures, due consideration shall be given to the question whether any particular programme or measure should apply to all, or a specified part, of the maritime area.

ARTICLE 4

1. In accordance with the penultimate recital of the Convention, no programme or measure concerning a question relating to the management of fisheries shall be adopted under this Annex. However where the Commission considers that action is desirable in relation to such a question, it shall draw that question to the attention of the authority or international body competent for that question. Where action within the competence of the Commission is desirable to complement or support action by those authorities or bodies, the Commission shall endeavour to cooperate with them.

2. Where the Commission considers that action under this Annex is desirable in relation to a question concerning maritime transport, it shall draw that question to the attention of the International Maritime Organisation. The Contracting Parties who are members of the International Maritime Organisation shall endeavour to cooperate within that Organisation in order to achieve an appropriate response, including in relevant cases that Organisation’s agreement to regional or local action, taking account of any guidelines developed by that Organisation on the designation of special areas, the identification of particularly sensitive areas or other matters.