Advisory Committee on the Marine Environment

ICES CM 2000/ACME:02
Ref.: ACFM + E

REPORT OF THE

WORKING GROUP ON ECOSYSTEM EFFECTS OF FISHING ACTIVITIES

ICES Headquarters
26 November–2 December 1999

International Council for the Exploration of the Sea

Conseil International pour l’Exploration de la Mer

Palægade 2–4 DK–1261 Copenhagen K Denmark

TABLE OF CONTENTS

Section **Page**

1 INTRODUCTION 1
 1.1 Venue and Attendance 1
 1.2 Terms of Reference 1

2 OVERVIEW AND GUIDE TO THE REPORT (Prepared by the Chair of WGECO) 2

3 REVIEW OF ECOSYSTEM MODELS AS BASIS FOR CHOOSING METRICS OF ECOSYSTEM STATUS AND EVALUATING INDIRECT EFFECTS OF FISHING 5
 3.1 Evaluating Ecosystem Effects of Fishing: Predictions from Ecosystem Dynamics Models 5
  3.1.1 Inventory of models of ecosystem dynamics 5
  3.1.2 Model type key 5
  3.1.3 Description of models and predictions for the ecosystem effects of fishing 6
  3.1.4 Detailed description of some models listed above 10
   3.1.4.1 A community metrics model—Huston's Dynamic Equilibrium Model 10
   3.1.4.2 Aggregate ecosystems models—ECOPATH, ECOSIM and ECOSPACE 13
   3.1.4.3 An age/size structured ecosystem model—European Regional Seas Ecosystem Model (ERSEM) 14
 3.2 Review of Recent Studies on the Direct Effects of Fishing. 14
 3.3 Development of Measures to Determine the Indirect Effects of Fishing 14
 3.4 Where does WGECO go from here? 15
 3.5 References 15

4 ECOSYSTEM EFFECTS OF FISHING ACTIVITIES IN THE BALTIC SEA 19
 4.1 Description of the Fisheries 19
 4.2 Effects on Target Species and the Food Web 20
  4.2.1 Effects due to by-catch and discarding 20
 4.3 Impact of Fishing Gear on Benthos 21
 4.4 Effects on Seabirds 21
 4.5 Effects on Marine Mammals 21
  4.5.1 Harbour Porpoises 21
  4.5.2 Seals 22
 4.6 Salmonid Reproduction Disturbances—M74 22
 4.7 Preventive Measures 22
 4.8 Summary 23
 4.9 Need for Further Research or Additional Data 23
 4.10 References 23

5 REVIEW PAST INFORMATION ON SECONDARY EFFECTS CAUSED BY DUMPING FISH REMNANTS IN THE BALTIC 24

6 BOTTOM TRAWL IMPACTS ON THE BENTHOS IN THE NORTH SEA AND IRISH SEA 25
 6.1 Approach to the Term of Reference 25
 6.2 Background Information from IMPACT II and the Scientific Literature 26
  6.2.1 Fleets, gears, and physical impacts 26
   6.2.1.1 Introductory remarks on gear types used in the report and fishing effort 26
   6.2.1.2 Size of bottom trawling fleets 27
   6.2.1.3 Fishing gears used by different fishing fleets 27
   6.2.1.4 Physical impacts 28
   6.2.1.5 Evidence in support of the effects of bottom trawls 29
   6.2.1.6 References 29
  6.2.2 Direct mortality 30
   6.2.2.1 Catch efficiency of commercial trawls 30
    6.2.2.1.1 Catch composition 30
    6.2.2.1.2 Catch efficiency for small-sized fish and invertebrates 31
   6.2.2.2 Direct mortality due to trawling 32
    6.2.2.2.1 Mortality of discards 32
    6.2.2.2.2 Damage of invertebrates 32
    6.2.2.2.3 Total direct mortality of invertebrates 33
   6.2.2.3 Scavenger responses to trawling 34
   6.2.2.4 General discussion: fishing mortality in invertebrate populations due to different types of trawl fisheries in the Dutch sector of the North Sea in 1994 35

TABLE OF CONTENTS

Section **Page**

6.2.2.5 Evidence in support of the effects of bottom trawls 35
6.2.2.5.1 Species composition..36
6.2.2.5.2 References..37
6.2.3 Comparison of disturbed and undisturbed areas..38
6.2.3.1 Introduction..38
6.2.3.2 Loch Gareloch 38
6.2.3.3 West Gamma wreck study 39
6.2.3.4 Iron Man and 41 Fathom Fast..40
6.2.3.5 General conclusions from the IMPACT II report on Chapter 3.7 (p. 370) 41
6.2.3.6 Generic potential effects supported by examples of 'Disturbed versus undisturbed areas' 42
6.2.3.6.1 Habitats..42
6.2.3.6.2 Species..43
6.2.3.7 References..44
6.2.4 Long-term trends in demersal fish and benthic invertebrates..45
6.2.4.1 Review of the IMPACT II report..45
6.2.4.1.1 Historical and recent data on epifauna in the southern North Sea (IMPACT II, Chapter 3.8.1) 46
6.2.4.1.2 Historical and recent data on macrofauna in the German Bight (IMPACT II, Chapter 3.8.2) 46
6.2.4.1.3 Long-term impact of bottom fisheries on several by-catch species of demersal fish and benthic invertebrates in the southeastern North Sea (IMPACT II, Chapter 3.8.3) 47
6.2.4.1.4 Shifts in the benthic community of the southeastern North Sea during extensive bottom trawl fishery (IMPACT II, Chapter 3.8.4)........................48
6.2.4.1.5 Long-term fluctuations in fish recruit abundance in the western Wadden Sea in relation to variation in the marine environment (IMPACT II, Chapter 3.8.5) 48
6.2.4.1.6 Abundance of dab and grey gurnard and trawlable biomass in relation to fishing effort (IMPACT II, Chapter 3.8.6)..48
6.2.4.1.7 Trends in abundance and length of eight target and non-target fish species in the North Sea (IMPACT II, Chapter 3.8.7)..49
6.2.4.2 Discussion of other long-term studies..49
6.2.4.2.1 Habitat change 49
6.2.4.2.2 Species composition..50
6.2.4.3 Other potential causative factors..54
6.2.4.4 Limitations of time series data studies..54
6.2.4.5 References..55
6.2.5 Summary..59
6.3 Potential Effects of Bottom Trawls..60
6.3.1 Introduction 60
6.3.2 Habitats..61
6.3.3 Species..62
6.3.4 Food-web and ecosystem properties..63
6.3.4.1 References..65
6.4 Mitigating the Effects of Bottom Trawls..66
6.4.1 Gear substitution..66
6.4.2 Gear usage 66
6.4.3 Gear modification..67
6.4.3.1 Lighten the gear 67
6.4.3.2 Use novel techniques 67
6.4.3.3 Develop more selective gears 68
6.4.4 Spatial closures..68
6.4.5 Real time closures..68
6.4.6 Improve habitat..68
6.4.7 Species adjustment 68
6.4.8 By-catch quota..69
6.4.9 References 69

TABLE OF CONTENTS

Section **Page**

6.5 Achieving reductions in effects 69
6.5.1 Context 69
6.5.2 Priority management measures 70
6.5.2.1 Major reduction in fishing effort 70
6.5.2.2 Closed areas 70
6.5.2.3 Gear substitution 70
6.5.2.4 Gear modification 71
6.5.2.5 Habitat rehabilitation 71
6.5.3 Governance changes 71
6.5.4 Specific immediate actions 71
6.5.4.1 Prevent expansion of areas impacted by bottom trawls 72
6.5.4.2 Prevent expansion of the numbers of bottom trawlers 72
6.5.4.3 Strengthen interactions with groups working on conservation of these ecosystems 72
6.5.4.4 Improve ability to detect and measure impacts 72
6.5.5 Reference 72

7 ECOSYSTEM MANAGEMENT OBJECTIVES 72
7.1 Introduction 72
7.2 Population and Species Reference Points/Objectives 74
7.2.1 Populations of target and non-target species 74
7.2.2 Spatial properties 75
7.2.3 Dependent species 75
7.2.4 Scavenger caused effects 75
7.3 Habitat Features 76
7.4 Genetic Properties of Populations 77
7.5 Emergent Properties of Ecosystems 78
7.5.1 Emergent properties: What are they? 78
7.5.1.1 Does fishing put emergent properties at risk? 78
7.6 Reference Points for Management 78
7.6.1 Populations and species 78
7.6.1.1 Direct mortality 78
7.6.1.2 Range 79
7.6.1.3 Ecologically dependent species 80
7.6.1.4 Scavengers 80
7.6.2 Habitats 81
7.6.2.1 Criteria for selection 81
7.6.2.2 Possible reference points 81
7.6.3 Genetic properties 82
7.6.4 Emergent properties 82
7.7 Conclusions and Way Forward 83
7.8 References 84

8 FOOD FOR THOUGHT 86
8.1 Social and economic aspects of ecosystem effects of fishing 86
8.2 Research proposal: Biodiversity of fish communities—towards testable hypotheses 86
8.2.1 Introduction 86
8.2.2 Approach 87
8.3 References 88

9 FUTURE ACTIVITIES FOR WGECO AND RELATED ICES GROUPS 88
9.1 Meeting / Workshop on Ecosystem Models 88
9.2 Ecosystem Effects of Deep-water Fisheries 88
9.3 Testable Hypotheses about Fishing Effects on Biodiversity 88
9.4 Liaise with the Study Group on Marine Habitat Mapping 89
9.5 Quantifying and Controlling Effect 89

ANNEX 1: LIST OF PARTICIPANTS 90

ANNEX 2: LIST OF WORKING DOCUMENTS 93

1 INTRODUCTION

1.1 Venue and Attendance

The Working Group on Ecosystem Effects of Fishing Activities (WGECO) met from 22 November through 1 December 1999 at ICES Headquarters in Copenhagen. The meeting was opened at 10.00 hrs by the Chair, Dr Jake Rice, and WGECO was welcomed by Dr Janet Pawlak, the ICES Environment Adviser. Members attending the meeting were:

Magda Bergman	Netherlands
Jean Boucher	France
Jeremy Collie	Denmark
Niels Daan	Netherlands
Martin Dorn	USA
Siegfried Ehrich	Germany
Arill Engås	Norway
Dick Ferro	UK (Scotland)
Alain Fréchet	Canada
Chris Frid	UK (England)
Simon Greenstreet	UK (Scotland)
Sture Hansson	Sweden
Louize Hill	Portugal
Eric Hoffmann	Denmark
Ellen Kenchington	Canada
Knut Korsbrekke	Norway
Ronald Lanters	Netherlands
Brian MacKenzie	Denmark
Philip MacMullen	UK (England)
Eyfinn Magnussen	Faroe Islands
Livingston Marshall	USA
Stefan Akí Ragnarsson	Iceland
Jake Rice (Chair)	Canada
Stuart Rogers	UK (England)
Francisco Sanchez	Spain
Mark Tasker	UK (Scotland)

Contact information for the participants is attached as Annex 1.

1.2 Terms of Reference

ICES C.Res. 1999/ACME:2

The **Working Group on Ecosystem Effects of Fishing Activities** [WGECO] (Chair: Dr J. Rice, Canada) will meet at ICES Headquarters from 22 November to 1 December 1999 to:

a) commence a review of the framework in which ICES evaluates and advises on ecosystem effects of fishing. This review would include, *inter alia*, to:
    i. review the principal models of ecosystem dynamics and develop specific predictions based on each of them for the ecosystem effects of fishing,
    ii. provide a synthesis of the findings of recent studies on the direct effects of fishing on marine ecosystems and critically assess the possible indirect influences of fishing on marine ecosystem function with a consideration of current levels of understanding of them,
    iii. formulate, based on i) and ii), suggestions of appropriate areas for the development of measures of the indirect effects of fishing on marine ecosystems;

b) review information on ecosystem effects of fishing activities in the Baltic Sea, as contained in the 1997 ACME report, and update this material (by April 2000) as a contribution to a chapter on ‘Marine fish migratory and freshwater species in the Baltic Sea area’ for the HELCOM Fourth Periodic Assessment of the State of the Marine Environment of the Baltic Sea, 1994–1998 [HELCOM 2000/3];

c) review past information provided by ICES on possible secondary effects caused by dumping fish remnants, in the context of its applicability to the Baltic Sea, and update as necessary (by April 2000) [HELCOM 2000/5];

d) consider the report ‘The effects of different types of fisheries on the North Sea and Irish Sea benthic ecosystems’ (Lindeboom and de Groot, eds.) and suggest and justify possible management actions by which the effects of gears discussed in the report on benthic communities could be measurably reduced, without unduly reducing the possibilities of catching commercially important species; consider all possibilities, such as establishing closed areas for bottom gears, reducing the weight of bottom gears, etc. [Request from EC 15–09–98];

e) begin consideration of the development of integrated management objectives as a basis for an ecosystem approach to management, integrating fisheries and environmental aspects.

WGECO will report to ACFM intersessionally, and to ACME in January 2000. WGECO will report to the Marine Habitat and Resource Management Committees at the Annual Science Conference in September 2000.

2 OVERVIEW AND GUIDE TO THE REPORT (Prepared by the Chair of WGECO)

This meeting of WGECO was again characterized by a diverse and hard working group of participants. As with the meeting in 1997, time was very unevenly distributed among Terms of Reference. In particular, Term of Reference d), reported in Section 6, dominated the workload, as it does coverage in this report. This was necessary because of the importance of the advisory question associated with that Term of Reference. However, again there was frustration that only modest progress was made on Term of Reference a), to explore and document the strengths and weaknesses of various ecosystem models and approaches. We stress again that this work is essential in the medium term, if ICES is to provide sound advice on ecosystem issues.

At present there are many ecosystem models on the market, and even more methods for summarizing various properties of ecosystems. These models are inventoried and organized by approach in Section 3.1, as analysis methods and metrics were in Section 3 of the 1997 report (ICES, 1998a). However, again we were unable to carry out the step of actually testing the limits and validity of each of the models, to clarify which ones should be used as a basis for providing advice of which sorts. Our discipline remains in a situation where as long as someone has developed an ecosystem model (or metric), had it published, and advocates its use, it may be brought forward as a basis for advice, without a good understanding of the model’s (or metric’s) true information content and limitations. This situation would never be acceptable in the simpler cases of single-species stock assessments, and it surprises many of us that much lower standards for testing and verifying ecosystem models and metrics seem to be accepted by not just ICES, but the scientific community interested in ecosystem questions. We outline a way forward in Section 3.4, but progress on this important issue will require a different setting from an ICES Working Group trying to discharge other important Terms of Reference in support of sensitive and urgent advisory tasks. It should be noted that three specific ecosystem models are described in some detail in Section 3.2. This is **not** intended to be a specific endorsement of those models, but rather was included to take advantage of expertise present at the meeting by getting the information documented for future consideration, as was done for a few other models in Section 10 of the report of the 1997 meeting.

The two Terms of Reference on Baltic issues, b) and c), were handled well. It was regrettable that few experts from countries around the Baltic were able to attend the meeting, but those that did attend served us, ICES, and those interested in conservation of the Baltic, well. WGECO stresses, though, that if Baltic countries and agencies are interested in increasing the advice they receive on effects of fishing on the Baltic ecosystem, greater support for research on those themes is essential.

Term of Reference d) on trawl impacts on benthic communities in the North Sea is addressed in Section 6 of the Report. The Term of Reference was complex and sensitive, and discharging it required many hours of deliberation in plenary, and even more time working individually and in subgroups on different portions of the task. The diversity of expertise attending the meeting strengthened the quality of the results of our work, but also made achievement of consensus on issues a painstaking and gradual process. Disciplinary differences of viewpoint had to be reconciled at almost every step, and each discussion leading to consensus increased everyone’s appreciation of each aspect of this important conservation concern. For example, by having some experts on gear technology participate in the meeting, fisheries scientists and ecologists learned a great deal more about the diversity of practice in what is often called ‘a fishery’. The gear experts, who, not surprisingly, were more optimistic than many of the other participants about the potential for gear modifications to reduce or eliminate undesirable effects of bottom gears, also gained a better appreciation of a number of benthic concerns.

Progress was also made difficult by the pervasiveness of ill-defined terminology in this field. This shows up in areas such as the circularity of labeling species as ‘fragile’ and ‘robust’, in the use of a phrase like ‘bottom trawl’ to apply sometimes to a very limited number of gears used in the southern North Sea and other times to anything mobile which touches the seabed, and disciplinary differences in the nuances conveyed in many phrases. We have done our best to be extremely careful to always use the same words to mean the same thing, and explain clearly what we mean by terms and phrases with different interpretations in different regions or disciplines. It is unlikely that we could have a perfect record for navigating through the terminological minefield, however. It has also meant that sometimes phrases, sentences or even entire paragraphs are repeated in different parts of the report. This redundancy was thought necessary, because pieces of past reports have been extracted and used in a variety of ways. Where we felt that such an extraction may occur with this report, we took some pains to ensure that selective extraction would not distort our intent by leaving out important qualifying text presented in some preceding part of the report.

Section 6 is very long, to allow full treatment of all the components of the request. Because of the length, there was a need to have terse summaries of most conclusions, their supporting evidence, and their implications. Two key tables, Tables 6.2.5.1 and 6.4.4.1, in particular, contain a great deal of information in extremely distilled form. This made them very difficult to achieve consensus on, and it is a credit to all participants that individuals covering the full diversity of opinions on these issues were willing to cotinue working on the table until we could bring something this concise forward. The degree to which those tables distill very long and complex discussions also makes them very vulnerable to being misinterpreted or, even worse, misrepresented. Both because of the length and complexity of this section, and the danger of taking points out of context, we prepared Section 6.1 as a guide to exactly how to use the material which follows. **It is essential that all critical readers go through Section 6.1 carefully,** in order to understand how to use what follows. The presentation strategy explained in Section 6.1 should be kept clearly in mind when reading individual parts of Section 6.

Section 6.2 is quite long, summarizing and critiquing both the studies reported in the IMPACT II report and other literature directly relevant to addressing the request for advice. Following the format of the IMPACT II report, this section is divided into subsections on fleets, gears, and impacts of gears on habitats; direct mortality caused by bottom gears; studies comparing trawled and untrawled sites; and long-term studies. Despite our best efforts, because these subsections were prepared by different subgroups, there are minor differences in format and presentation. As far as possible the differences were smoothed out during review of the text in plenary, but minor differences remain. However, we are confident that the necessary information can be found readily in each subsection, and the minor differences in format are preferable to attempting major reorganizations of material on the last day of the meeting.

Each subsection ends by evaluating the soundness and extent of evidence for each of a list of possible impacts of bottom gears on benthic communities. This list of four physical and nine biological *possible* impacts was prepared early in the Working Group meeting. It was considered by the Working Group to cover fully the possible impacts that *might* be caused by bottom gears. It structures not just the concluding portion of each subsection of Section 6.2, but all of Section 6.3 and much of Section 6.4. Ranking the possible effects presented another challenge to WGECO, although it was easy to agree on the criteria for ranking—temporal scale, spatial scale, and direction of change. We call special attention to the fact that several effects may be given the same rank, with regard to our degree of concern that if they occur, they require mitigative action. Several types of biological effects and the smallest scale physical effects were not considered serious enough to justify major changes to fisheries, even if there was evidence that they might be occurring. Also, in Section 6.3.4 we discuss the issue of effects of these gears on food webs and emergent ecosystem properties. We conclude that evidence for such effects is far less compelling than for direct effects, and that if indirect effects are occurring, measures to mitigate the direct effects which are documented well will move these systems in the direction of improved conservation of the more emergent properties as well.

Similarly, a list of the types of possible remedial actions for the effects was developed over several plenary discussions. Reconciling differences in opinion regarding the effectiveness of each class of measure for each class of effect was particularly challenging. WGECO stresses that the information in Table 6.4.4.1 should be viewed very much ‘at arm’s length’. Relative rankings of effectiveness are for a measure in the class that is well designed and well implemented. Regardless of how high the general ranking, any measure, done poorly, may provide no mitigative advantage. Moreover, at least some of the measures we class as having generally low effectiveness for a particular effect might be beneficial for some restricted and well-targeted applications.

Only after WGECO had reached consensus on the lists and cross-tabulation of effects and classes of mitigative actions, did we tackle the portion of the request asking for recommendations of specific actions to reduce documented effects on the North Sea and Baltic Sea. The first attempt resulted in Section 6.5.2, where we present our priority management measures. However, on reviewing that section it was clear that all the entries were general actions, and managers would require additional problem-specific advice before actions could be taken. Provision of the additional advice would, in turn, require substantial additional scientific information and, particularly in the case of governance changes, long and probably difficult consultation with a wide range of interests. However, WGECO was able to go further once that basis was in place, and Section 6.5.4 presents four actions that managers could take immediately, without additional scientific advice and without unduly restricting the opportunity to catch fish, if they are serious about reducing undesirable and unnecessary effects of fishing on benthic ecosystems.

Although the job is not done, WGECO made substantial progress on Term of Reference e) regarding ecosystem management objectives. WGECO attacked the problem by posing and attempting to answer (from the literature and critical dialogue) four questions about each class of ecosystem objective identified in the 1997 WGECO report. The questions were:

* What ecosystem properties are at risk?

* How do fisheries put them at risk?

* What can be measured that reflects directly how fisheries put the properties at risk?

* What value(s) of the measurement(s) should trigger management action?

Not surprisingly, the answers we found become less complete as one goes down the list. The fourth question, in particular, can only be answered in the context of specific systems. However, although specific candidate objectives and reference points are not proposed, the properties of operational objectives for specific applications are developed quite fully. This is most complete for ecosystem objectives of species and populations, and somewhat less so for habitats and population genetics reference points. For species, populations, and habitats, this material should be sufficient to guide the determination of operational objectives and associated reference points for specific applications. For genetics, the knowledge base may not yet be adequate to make objectives and reference points operational for most species and populations, but the direction in which investigations should go is made clear in the report.

When confronting objectives and reference points for emergent properties of ecosystems, WGECO again concluded that there is no scientific evidence at this time that such reference points are necessary, nor would they be practical. This conclusion crops up in Sections 3 and 6 of the report as well, from different lines of inquiry. It is important that this point be understood as WGECO intends. We have not concluded that all marine ecosystems are in good condition when viewed from the distance implicit in reporting their emergent properties. Rather, we have concluded that there is no evidence that emergent properties would be at risk, if the fisheries were sustainable when measured by impacts on target and non-target species and habitats. These latter tasks are ones that the scientific and management communities should know how to do—certainly if they don’t know how to conserve species and habitats, it is unrealistic to argue that they would know how to conserve more abstract properties like biodiversity or trophic structure. Management efforts directed at ensuring conservation of species and habitats are easier to develop, implement, enforce, and monitor, and if this were truly done seriously, all the credible scientific evidence suggests that higher order properties of ecosystem would be conserved as well.

The report ends with two sections looking into possible future directions for WGECO. The Food for Thought Section develops two potential research lines arising out of discussions during the preparation of the response to Term of Reference d). The first is to come to grips with the social aspects of fisheries. It notes that ICES has begun to build links to the research community studying the economics of fisheries, so that ICES can treat the economic consequences of biological advice, and economic constraints on managers’ actions to implement the advice, as research issues themselves. The section stresses, though, that many constraints on the capacity of managers to implement biological advice have social, rather than strictly economic, roots and these questions are not being studied with comparable seriousness. The second part of Section 8 develops a research approach to addressing the circularity inherent in much of the literature characterizing the types of species likely be affected strongly or weakly by fisheries. It is a problem tractable with information currently available, and would show significant returns in improved advice on species’ and ecosystems’ responses to fishing, if resources were allocated to pursuing the proposed research.

Section 9 of the report summarizes five specific activities for WGECO, each arising from earlier portions of the WGECO report. It was clear to participants that WGECO cannot simultaneously pursue all five lines with equal intensity of focus. We call these different potential activities to the attention of the science and advisory committees to which we report, and encourage them to consider which activities should be given the highest priority.

3 REVIEW OF ECOSYSTEM MODELS AS BASIS FOR CHOOSING METRICS OF ECOSYSTEM STATUS AND EVALUATING INDIRECT EFFECTS OF FISHING

3.1 Evaluating Ecosystem Effects of Fishing: Predictions from Ecosystem Dynamics Models

For the first part of this term of reference, WGECO was asked to commence a review of the principal models of ecosystem dynamics and develop specific predictions based on each of them for the ecosystem effects of fishing;

Due to the higher priority of other terms of reference during this meeting, this section is a progress report and not a completion of the term of reference.

3.1.1 Inventory of models of ecosystem dynamics

There has been a multitude of models constructed, each of which purports to illustrate the dynamics of ecosystems. A useful classification of these models was provided by a flowchart in Hollowed *et al.* (in press). See Figure 3.1.1. This flow chart provides a useful guide to grouping together models based on similar constructs, requiring similar input variable data and producing similar output predictions. We have used this flow chart to produce the key to ecosystem models presented in Section 3.1.2. We have used the key to assign models gleaned from the theoretical ecology and fisheries science literature to various categories or ‘families’ of models. Models within a family will provide essentially the same sort of insight into how fishing may affect the ecosystem. It may be that each model may require slightly different input data, thus some models within a family may be more or less appropriate, or practical, when attempting to address a particular issue. Different families of models will essentially address different issues, or provide different insights into ecosystem operation. In earlier reports we have concluded that there is no evidence that we will fail to safeguard the marine ecosystems as long as single-species issues are addressed adequately, such that no one component or species within the ecosystem is subject to unsustainable mortality. This message is confirmed in our current deliberations. Only a few of the models mentioned below consider single species in a way that can be directly compared with current assessment models, dealing as they do in the most part with multispecies interactions. Multispecies models therefore provide a means of examining how fishing disturbance might affect the emergent properties of ecosystems, in particular food-web dynamics and change in species diversity. This, in turn, might provide the definitive answer to questions about the risk posed to emergent ecosystem properties in systems where individual species are not at risk.

In Section 3.1.3, we briefly describe some of the models in each of the model families defined by the key. We indicate what they do, some of the key assumptions, and the type of output they provide. In Section 3.1.4, we describe a few of the models in much greater detail. We describe their data input requirements and determine how their output might provide insight as to how variation in fishing pressure might affect the output parameters considered.

3.1.2 Model type key

1) Is habitat suitability explicitly included?
   No go to 2
   Yes Basin model (MacCall, 1990)

2) Is model based on community metrics?
   No go to 3
   Yes Dynamic equilibrium model (Huston, 1979, 1994)
   Size-spectrum model (Pope *et al.*, 1988)

3) Predator-prey-feedback included?
   No Prey added to single-species models (Furness, 1978)
   Predators added to single-species models (Livingston and Methot, 1998)
   Yes go to 4

4) Are the environment and lower trophic levels included?
   No go to 5
   Yes go to 7

5) Age structure?
   No Multispecies production models
      a. Multispecies Lotka-Volterra models (May *et al*., 1979)
      b. Predator-prey models with non-linear interactions (Collie and Spencer, 1994; Spencer and Collie, 1996; Basson and Fogarty, 1997)
      c. Spatially-explicit predator-prey models (Pascual and Levin, 1999)
   Yes go to 6

6) Dynamic multispecies models with age-structure [a], variable growth [g], multiple fleets [f], spatial structure [s]
   a. MSVPA (Sparre, 1991) [a, f]
   b. MSGVPA (Gislason, 1999) [a, g, f]
   c. Length-based MSVPA (Dobby *et al*., 1999) [g, f]
   d. MULTSPEC (Tjelmeland and Bogstad, 1998) [a, f, g, s]
   e. Bormicon (Stefànsson and Pàlsson, 1997) [a, f, g, s]
   f. 4M (Vinter and Thomsen, 1998) [a, f, g, s]

7) Age-structured?
   No go to 8
   Yes go to 9

8) Aggregate system models with time dynamics [t], spatial dynamics [s]
   a. N→P→Z models
   b. ECOPATH (Polovina, 1984)
   c. ECOSIM (Walters *et al*., 1997) [t]
   d. ECOSPACE (Walters *et al*., 1998) [t,s]

9) Age/size structured ecosystem models
   a. Andersen and Ursin (1977) North Sea model [t][*]
   b. ERSEM (Baretta *et al*., 1995) [t, s]
   c. Individual based models (Murphy *et al*., 1998; Batchelder and Williams, 1995)
   d. OSMOSE (Shin and Cury, 1999) [a, t, s]

3.1.3 Description of models and predictions for the ecosystem effects of fishing

The classes of models correspond to the key above. We list the principal properties of each class of model without reviewing their validity or usefulness. Generic predictions are made about the ecosystem effects of fishing if each of the models were a correct description of the ecosystem. Several of the models have been described more fully in Section 3.1.4. A more thorough review will require a more complete description of each model's properties, consideration of the underlying assumptions or theory, and empirical evidence for the model. For those models that have been implemented and parameterised, estimates of the ecosystem effects of fishing can be made.

Habitat-based models

Properties

The population growth rate depends on habitat suitability. Realised suitability is a function of local population density. Individuals distribute themselves with an ideal-free distribution such that realised suitability is equal over the entire range of habitats (see MacCall, 1990).

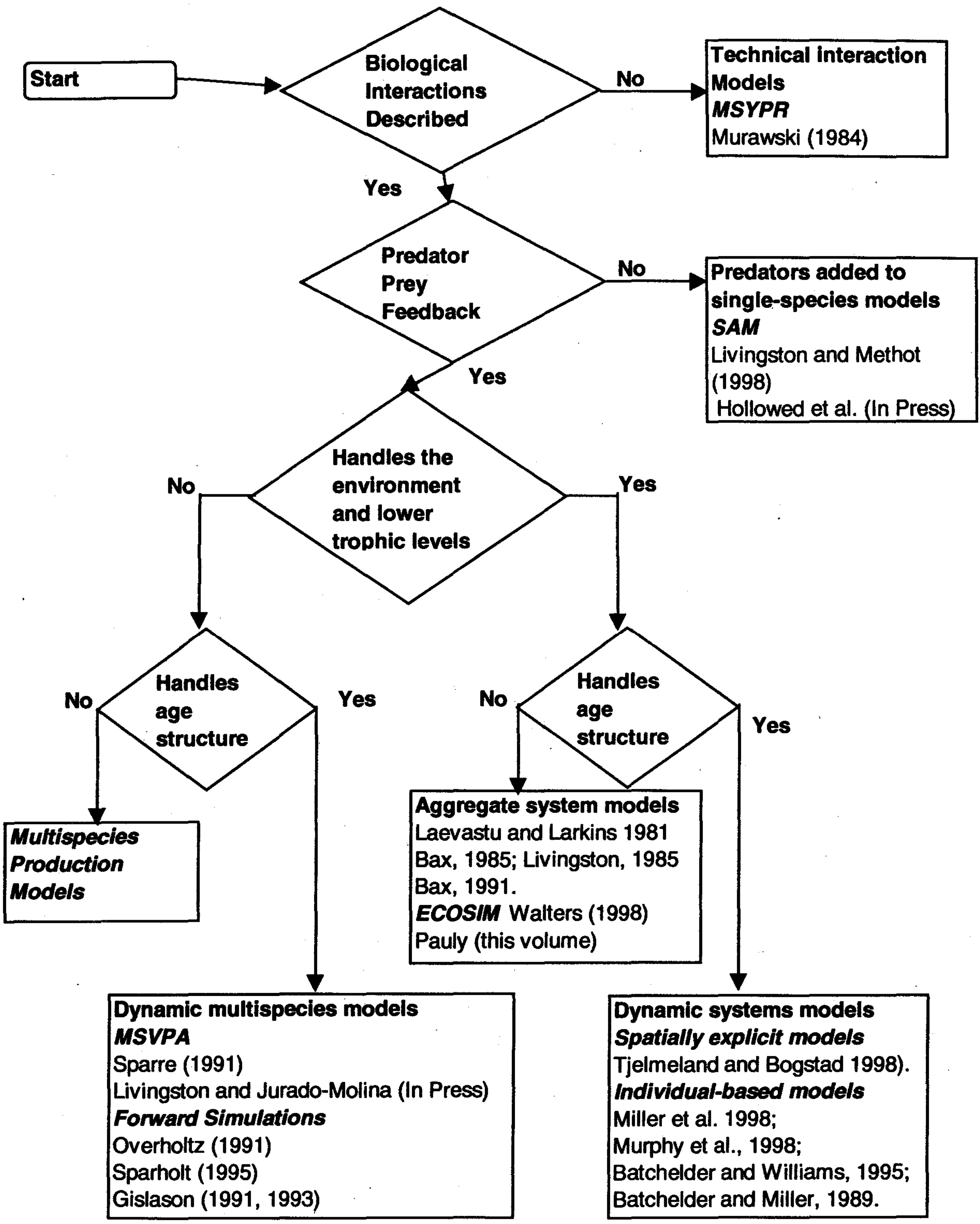
Predictions

As fishing reduces the total population size, the population range will contract to the most suitable habitats. Catchability will remain constant in the core habitat as total population size declines because local population density is highest in the most suitable habitats.

---

[*] this model is no longer in use

Figure 3.1.1. Flow chart summarizing classification of multispecies models. **Bold** letters indicate model classification, *italicized* letters indicate sub-categories of models. References for classes and sub-categories of models are provided in the text of the paper by Hollowed *et al.* (in press).



Models based on community metrics

Properties

These models are generally applied at the community level and do not describe the abundance of individual species from one generation to the next. According to the dynamic-equilibrium model, species diversity is a function of ecosystem productivity and natural disturbance levels (see more detailed description of this model in Section 3.1.4.1). The size-spectrum model describes the decrease in number of individuals with increasing length classes.

Predictions

The dynamic equilibrium model predicts that species diversity will change in response to fishing disturbance. The direction and magnitude of the change depends on levels of production and natural disturbance in the unimpacted ecosystem. The slope of the size spectrum is expected to become steeper when the community is exploited. The relationship between the slope of the size spectrum and fishing mortality has been investigated by simulation (Gislason and Lassen, 1997) and by comparing the size spectra from fish communities around the world (Bianchi *et al*., in press).

Single-species models with variable prey or predators

Properties

These are dynamic single-species models with either variable prey abundance (e.g., fish abundance for seabirds) or variable predator abundance (e.g., predators on fish populations). Trophic feedback is in one direction only.

Predictions

In the case of variable prey, harvesting the prey species (forage fish) will reduce the reproductive rate or growth rate of the predators (seabirds). In the case of variable predation, harvesting the predator will reduce the natural mortality of the prey species and may lead to higher prey recruitment (Walters *et al*., 1986). Harvesting the prey would not affect the predators because of the lack of prey-predator feedback. These models could perhaps be used to examine some of the immediate effects of alteration in the abundance of particular single species, i.e., the type of change which has commanded most attention throughout the remainder of the report.

Multispecies production models

Properties

The simplest examples derive from the Lotka-Volterra model and have first-order linear interaction terms. Density-dependent regulation of the prey is necessary for system stability.

Predictions

Harvesting the predator increases prey abundance and hence the potential prey yield (May *et al*., 1979). Harvesting the prey reduces predator abundance and the potential predator yield.

Predator-prey models with non-linear interactions

Properties

Non-linear functional responses and time delays in the effects of predation can give rise to multiple equilibrium levels of predators and prey populations. With a type-two predator functional response, there may be two equilibria: a stable upper equilibrium and a lower unstable equilibrium. With a type-three predator functional response, there may be two stable equilibria separated by an unstable equilibrium (Steele and Henderson, 1981). Habitat suitablity can be linked to species carrying capacity (Sainsbury, 1991).

Predictions

The effects of fishing are generally the same as for other multispecies production models (see above). In addition, incremental changes in either the predator or prey exploitation rates can cause the system to flip rapidly from one equilibrium level to the other (Collie and Spencer, 1994; Spencer and Collie, 1996). Prey populations can get trapped in ‘predator pits’ such that recovery may be delayed and require substantial reductions in fishing mortality. Fishing can reduce habitat suitability and selectively reduce the abundance of species which depend on that feature of the habitat (Sainsbury, 1991).

Dynamic spatially structured multispecies models

Properties

Predator-prey dynamics are simulated at small spatial scales. The spatial structure may include prey refugia. The dynamics of the populations can be examined by aggregating the spatial grid into larger ‘windows.’ Such models can exhibit oscillatory behaviour and multiple equilibria when examined on intermediate spatial scales.

Predictions

Spatial models of this type generally have not explicity included fishing, but fishing mortality could be added.

Dynamic age/size-structured multispecies models

Properties

These models track the fates of cohorts backward in time (MSVPA, MSGVPA, 4M) or forward from recruitment (Multspec, Bormicon). Natural mortality is a dynamic function of predator abundance and prey availability. Length-based MSVPA removes the dependence on age-length keys by incorporating growth models. In the models with variable growth rates, size at age is a function of food availability, as observed in boreal ecosystems. Models with spatial structure explicitly consider the seasonal overlap between predators and their prey.

Predictions

Harvesting the predators reduces natural mortality of the prey species, resulting in increased prey abundance and increased potential prey yields. In models with variable growth, harvesting the prey will reduce predator growth rates, resulting in reduced predator biomass and potential predator yields. The indirect effects of harvesting one species on other species in the community may be positive or negative depending on the food web structure. Spatially explicit models allow for local depletion of prey abundance (‘understocking’) even when total prey abundance remains undepleted.

Aggregate ecosystem models

Properties

These models derive from food webs and energy budgets; the units may be energy, carbon or biomass. Species are aggregated into functional groups, especially at the lower trophic levels. The trophic interactions are generally first-order linear. Static mass-balance models (e.g., ECOPATH) assume that the production of each ecosystem component is balanced by losses due to predation, non-predation mortality and export. Dynamic ecosystem models require some form of density dependence to prevent species extinctions. In ECOSIM, this stabilisation is provided by partitioning the functional groups into available and unavailable pools (Walters *et al.*, 1997). Spatially explicit ecosystem models (e.g., ECOSPACE) can model prey availability explicitly and can also define habitat suitability.

Predictions

In static ecosystem models, fishing mortality must be balanced by increased production or reduced predation on that functional group. Dynamic ecosystem models can describe the time dynamics of these changes to a new steady state. In general, harvesting a functional group will reduce its own biomass and the biomass of those groups that feed on it, while increasing the biomass of those groups which are preyed on. These changes can propagate along the food chain, resulting in a trophic cascade (Carpenter *et al*., 1985).

Age/size-structured ecosystem models

Properties

This class of models is distinguished from the aggregate ecosystem models in that the individual functional groups are generally less aggregated and their dynamics are described with greater temporal resolution. Given the high level of detail, these models are often constructed as simulations and individual based models (e.g., Shin and Cury, 1999) as opposed to statistical models fit to data. Coupled biophysical models that incorporate flow fields concentrate on the lower trophic levels and planktonic life stages.

Predictions

Many of the existing models of this type do not include the upper trophic levels (i.e., fish) and therefore cannot be used to predict the effects of fishing in their present form.

3.1.4 Detailed description of some models listed above

The descriptions are intended to be illustrative, and are not meant to suggest that WGECO endorses these models over others which are not described in detail at this time.

3.1.4.1 A community metrics model—Huston’s Dynamic Equilibrium Model

Initial theoretical community ecology models presupposed that the species diversity of communities was underpinned by competitive exclusion until ultimately competitive equilibrium was achieved (MacArthur, 1970; MacArthur and Levins, 1967). Many early studies supported these ideas (e.g., Park, 1948, 1954; Pianka, 1973, 1975, 1983; Fenchel, 1975; Davidson, 1977), but not all. Communities were identified which, while apparently stable, were clearly not at competitive equilibrium (e.g., Paine, 1966; Patrick, 1975). These suggested that predation was the dominant process governing the structure of communities. In its absence communities progressed to competitive equilibrium, in its presence an alternative stable, or even unstable, state existed (e.g., Connell, 1975). The apparently opposing effects of competition and predation led Huston (1979) to propose the ‘dynamic equilibria model’ of species diversity, the fundamental predictions of which are encompassed in Figure 3.1.4.1.1. Huston considered predation to be simply one source of ‘disturbance’, i.e., a factor that caused mortality and limited the scope for population growth. In the application of this model, fishing can be considered as a disturbance resulting in mortality, causing at worst, reductions in population size, or at best, limiting the extent of population increase. Variation in productivity puts limits on the amount of food resources available to competing predators.

The theory assumes a unimodal response of species diversity to both disturbance and productivity. Thus for example, nutrient enrichment events have been shown to result in both increases and decreases in species diversity (e.g., Mirza and Gray, 1981; Russo, 1982), i.e., both limbs of the unimodal relationship. At low disturbance frequency, diversity is highest at the intermediate to low end of the productivity spectrum. At the lowest productivity, population growth is so slow that extinction is a distinct possibility. At higher productivity, the scope for growth is sufficiently high that the populations of the dominant competitors will be able grow fast enough, so as to be able to capture an increasingly unequal share of the resources. Growth of the subordinate competitors will be impeded and they will be excluded before the next disturbance occurs. At higher disturbance frequencies, diversity will be lower at low productivity levels; some populations will be unable to grow fast enough to recover sufficiently between disturbance events and they will be driven to extinction. At intermediate productivity levels, populations will be able to grow fast enough to avoid extinction, but the growth of the dominant competitors will be checked, so preventing competitive exclusion. Species diversity in areas of high disturbance frequency and high productivity will be lower than in areas of intermediate levels of both because the high frequency of disturbance is likely to eliminate *k*-selected species and so reduce the total species pool available. Huston (1994) examined a large number and variety of species assemblage data sets and generally found that variation in species diversity could be explained in terms of the dynamic equilibria model.

What insight can this model provide regarding the impact of fishing disturbance on North Sea groundfish species assemblages? Figure 3.1.4.1.2 shows that the relationship between diversity and disturbance is strongly influenced by environmental productivity. Consider an increase in disturbance over the range of 20 to 80 units at the three levels of productivity marked on Figure 3.1.4.1.1. In productive environments increasing disturbance results in increased species diversity, while in unproductive environments the opposite trend is observed. In moderately productive environments diversity increases as disturbance rate increases to intermediate levels, thereafter diversity decreases. Secondly, the model suggests that without accounting for variation in productivity in some sort of multivariate analysis, it could prove difficult to determine any relationship at all. Figure 3.1.4.1.3 looks at the mean (and standard deviation) species diversity predicted by the dynamic equilibrium model at various disturbance levels over the whole productivity range. The three levels of fishing disturbance shown on Figure 3.1.4.1.1 are indicated. This graph suggests that we would be very unlikely to detect any significant variation in species diversity among the three fishing disturbance regimes from sets of random samples taken across the full productivity range. Finally, Figure 3.1.4.1.4 examines the response of species diversity to variation in productivity at the three levels of fishing disturbance indicated on Figure 3.1.4.1.1. In low productivity environments, species diversity is highest at low disturbance and least at high disturbance, a ranking which intuition suggests might be expected. However, at intermediate productivity the ranking starts to alter, so that where productivity is high the order is completely reversed; diversity is greatest where disturbance is most prevalent and *vice versa*.

Figure 3.1.4.1.1. Huston's (1979) 'dynamic equilibria model' of species diversity.

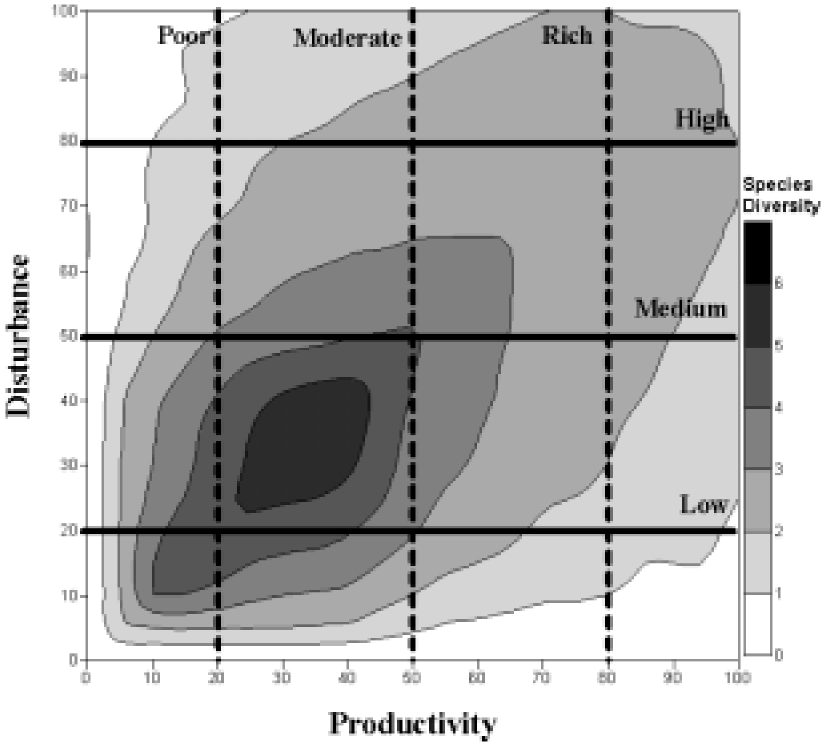


Figure 3.1.4.1.2. Relationship between species diversity and fish disturbance intensity at three levels of environmental productivity.

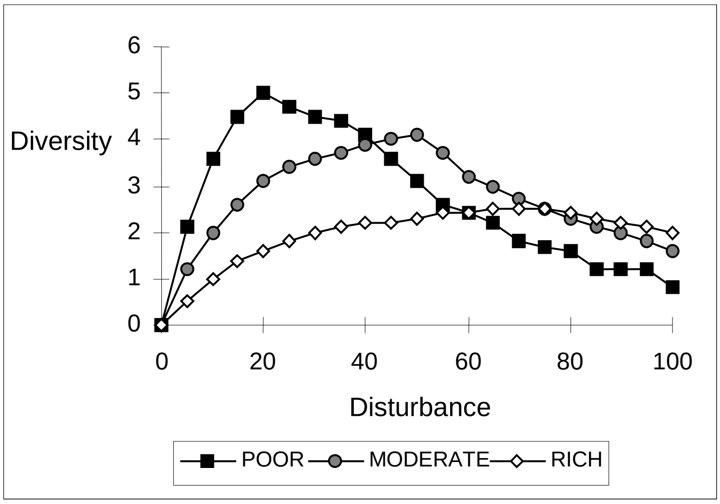


Figure 3.1.4.1.3. Effect of disturbance on the mean diversity and standard deviation estimated over the full productivity range. Low, medium, and high disturbance ranges shown on Figure 3.1.4.1.1 and plotted in Figure 3.1.4.1.2 are indicated.

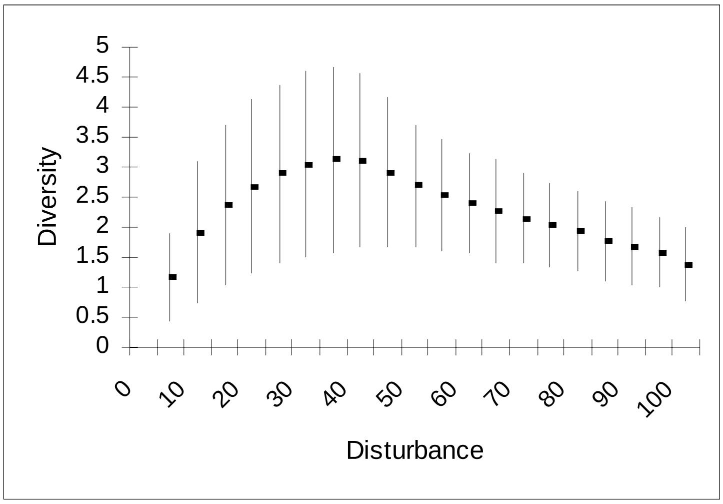
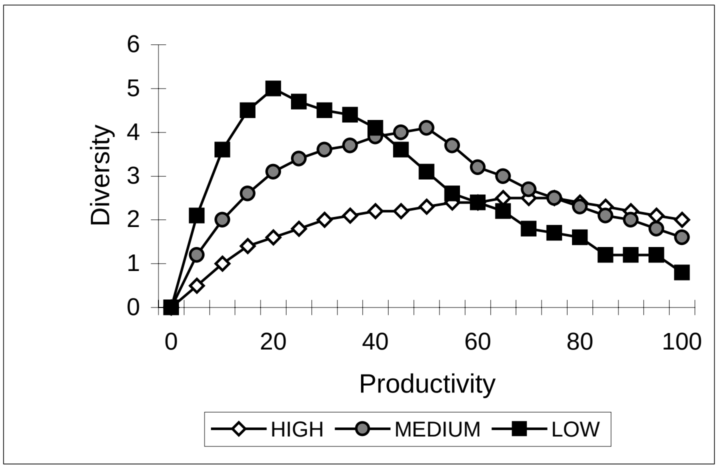


Figure 3.1.4.1.4. Relationship between species diversity and productivity at three fishing disturbance intensities.



The effects of disturbance on community species diversity are thus far from clear-cut (e.g., Death and Winterbourn, 1995). They may be particularly complex in marine ecosystems (Russo, 1982; Menge and Sutherland, 1987), underlining the necessity for the provision and validation of underlying theory. One of the difficulties in applying such models, however, is the frequent absence of the necessary data to fully parameterize it, in this case a direct measure of system productivity. One approach would be to find suitable correlates, such as water depth in marine systems (Rosenzweig and Abramsky, 1993). However, the use of such substitutes requires caution (Rosenzweig, 1995). A second approach may be to use empirical relationships (e.g., Brey, 1990; Duineveld *et al.*, 1991; also see Bryant *et al.*, 1995) to estimate benthic production from benthic standing crop biomass data. Such data may be available for the North Sea, at least in basic form, collected during the ICES North Sea Benthos Survey (Heip *et al.*, 1992; Basford *et al.*, 1993), and additional data continue to be collected. This raises the possibility of examining the relationships in spatial variation in fishing disturbance, benthic production and demersal fish species diversity to explore the value of Huston's model as a tool to enable us to predict the consequences of changes in fishing practice on groundfish species diversity.

3.1.4.2 Aggregate ecosystems models—ECOPATH, ECOSIM and ECOSPACE

All three of these aggregate ecosystem models were used in the study carried out by Sanchez and Olaso (1999), described in the working paper presented to WGECO. The use of these models in this study is summarised here by way of an illustration of how they might be applied to examining the indirect effects of fishing on marine ecosystems.

The study used ECOPATH to estimate the following: the trophic level of the fishery, the transfer efficiency between trophic levels, the mixed trophic impact with the fishery as both an impacted and an impacting component, and the historical mean trophic level of the fishery. The input data required were:

- An estimation of biomass (obtained from ICES, ICCAT and survey data in this study; when this value could not be estimated it was back calculated from the model using an alternative value for ecotrophic efficiency (EE) of 0.95);
- An estimation of mortality (obtained from ICES, ICCAT and survey data in this study);
- An estimation of consumption-to-biomass ratios and diet composition of each predator; and
- Landings and discard data from the fishing fleet.

Feeding habits form the links between groups derived from predator-prey relationships determined by stomach content analysis, or from literature values. Discards were considered and 60 % were returned to the model (the percent that falls to the seabed), and the 40 % taken by seabirds (not included in the model) were accounted for in the category ‘exports’. The model was parameterised as ‘top down’ such that flow at low trophic levels was set so as to match the food demands of the top levels. Achieving this energy balance assumes a steady-state equilibrium, and is perhaps one of the major drawbacks of the ECOPATH model.

Outputs: Major biomass flow diagram for the studied ecosystem for a given time, showing which groups determine the main flow and which groups form the link between pelagic and demersal groups. The flow of discards can be mapped, and the groups which are most impacted by these discards are identified. This model also provides insight to the direct and indirect trophic interactions within the ecosystem.

ECOSIM is a dynamic extension of ECOPATH (Walters *et al*., 1997). It includes biomass and size structure dynamics, and uses the same mass balance assumptions as ECOPATH for parameter estimation. The study by Sanchez and Olaso (1999) used ECOSIM to simulate the effect of different fishing regimes on the flow structure of the ecosystem.

Outputs: Data showing the evolution of biomass trends over time for trophic groups as a function of different fishing rates. The relationship between equilibrium biomasses at different fishing levels can thus be graphically displayed. These relationships can be predetermined as being ‘bottom up’, ‘top down’ or ‘mixed’ controlled systems. The predictions can also be plotted over time. This representation then shows how different species groups are affected by different levels of trawling and how some populations would decline and others could recover.

ECOSPACE is a mesoscale spatial simulation tool for predicting spatial patterns and runs the ECOPATH model through ECOSIM to model the response of the ecosystem (Walters *et al*., 1998). Sanchez and Olaso’s (1999) study used ECOSPACE to explore trophic and spatial relationships and to evaluate management options.

The input data required were:

ECOPATH model plus habitat-related parameters, thus replicating ECOSIM dynamics over a spatial grid of ‘homogeneous’ cells and linking the cells allowing for the dispersal of organisms and changes in fishing effort. ECOSPACE also allows for spatial variation in productivity and can represent habitat ‘preferences’ according to different criteria (differential dispersal, predation rates, feeding) (Walters *et al.* (1998)).

Outputs: A series of base maps illustrating the evolution of trophic group abundance in different situations. The example of Sanchez and Olaso (1999) illustrates the ecosystem response to a closed area instigated to protect hake recruits.

These three models provided predictions on the functioning of the ecosystem, and can provide predictions about ecosystem effects of each fishery type on different functional groups in the foodweb. This would provide an idea of the importance of each fishery for the trophic dynamics throughout the system.

3.1.4.3 An age/size-structured ecosystem model—European Regional Seas Ecosystem Model (ERSEM)

Most of the early models used to describe energy/carbon/nitrogen flow through marine food webs essentially assumed a steady-state dynamic equilibrium situation; they were ‘static’ models (e.g., Steele, 1974; Billen, 1978; Jones, 1982, 1984; Cohen *et al*., 1982; Mommaerts *et al*., 1984). The alternative view is that the food web is not at equilibrium and is instead constantly responding to environmental and/or anthropogenic forcing. Describing such situations required the application of ‘dynamic’ time-evolving models (Fransz and Verhagen, 1985; Billen and Lancelot, 1988). Both of these approaches require simplified representation of the ecosystem by the aggregation of species into functional groups. The degree of species aggregation varies considerably between models. An overview of many of these earlier models is provided by Fransz *et al*. (1991).

More recently many research institutes around the North Sea were involved in a project to develop a spatially explicit model of carbon pathways through the North Sea ecosystem (Baretta *et al*., 1995). The model used the ten ICES areas to achieve a relatively coarse spatial resolution. The five offshore boxes were split into two by depth to model the effect of the thermocline. Primary production occurred in the upper box, whilst the lower box included the links to the benthos. A general circulation model was used to generate daily horizontal exchange rates of dissolved and suspended constituents between the ten surface boxes. Vertical transport between the five upper and lower boxes was based on determined sinking rates for the sedimentation of particulates and turbulent diffusion for the dissolved constituents.

The physical model contained information specific to the area to be modelled, whereas the biological/chemical sub-models were constructed to be non-site-specific. The biological variables were represented as functional groups expressed as units of organic carbon and the chemical variables as internal pools in the biological variables and as dissolved inorganic pools in the water and the sediment, expressed in units of N, P, and Si.

The model is modular in its construction, with each module dealing with a related collection of functional groups, thus the Zoobenthos model included the functional groups: Benthic Carnivores; Suspension Feeders; Deposit Feeders; and Meiobenthos. The modules were set up to run within the ten ICES boxes and the five lower layer boxes using the Software Environment for Simulation and Analysis of Marine Ecosystems (SESAME) (Ruardij *et al*., 1995). The modules were linked to allow the exchange of carbon and nutrients between the modules. These were then routed through the functional groups comprising each module. Size and age structure was explicitly represented in the fish groups, but the remaining biological components were modelled as unstructured populations. This was deemed appropriate for taxa having short generation times relative to the annual cycle, and for taxa which did not cross more than one trophic level in their lifetime.

This model has not been used to examine the indirect effects of fishing on the emergent properties of the ecosystem. However, it would be a relatively easy step to introduce some rate of fishing mortality to the age-structured fish module. Such mortality could even be applied to take into account spatial and seasonal variation in the behaviour of different fisheries. The consequent changes arising from alteration to the carbon flow through the various modules could then be tracked over any given time period (1 to 10 years, or longer if appropriate). Examples of the sort of ecosystem food web alterations for which predictions could be obtained are: the consequences of sandeel fisheries operating in specific locations to top predators in the area, and to the zooplankton populations on which the sandeels prey. Furthermore, predictions could also be obtained regarding the effects of all these higher trophic level changes on the bentho-pelagic coupling in the system.

3.2 Review of Recent Studies on the Direct Effects of Fishing

Section 6 deals extensively with the direct effects of bottom trawling on benthic communities and that information is not repeated here. A complete review of the effects of fishing also needs to consider the effects on pelagic communities and the indirect effects of fishing.

3.3 Development of Measures to Determine the Indirect Effects of Fishing

WGECO considered that before any substantial progress could be made in addressing TOR 3iii, further progress with TOR 3i was necessary. Therefore, WGECO could not address this term of reference adequately in the time available.

3.4 Where does WGECO go from here?

For the second consecutive meeting, WGECO failed to resolve this Term of Reference, although the taxonomy of models is a major step towards bounding the task. There are two reasons for the slow progress on this Term of Reference.

First, Terms of Reference given to WGECO in support of ICES advisory tasks necessarily took priority, and WGECO simply could not focus on a set of questions that are known to be important, and which have challenged ecologists and modellers for decades. WGECO is certain that to make substantial progress on this Term of Reference, it will be necessary to have a meeting where it can focus exclusively on this set of activities. We suggest a special meeting of WGECO, of perhaps a week, to address only the testing and contrasting of different ecosystem models. The work would address the types of predictions that each type of model can make about effects of fishing on ecosystems, the ability of those predictions to be validated in scientifically rigorous ways, and the ability to gain useful insights for improved management advice from the models.

If WGECO does have such a meeting, the second important consideration is that it must be much better prepared to actually use a variety of ecosystem models on the same data sets, and to challenge each model with multiple data sets. The greatest insight will come when the data sets are from ecosystems with contrasting properties (for example, degree of connectedness, numbers of predators and prey, severity and frequency of major environmental perturbations) and are augmented by some simulated data sets where the true magnitudes of those factors are known. The former types of data sets will explore the ability of competing models to handle data from real ecosystems; the latter will explore whether a model’s predictions are reasonably reliable or badly misleading. Both the models and the data sets have to be ready on the first day of the meeting, so extensive planning will be essential to the success of the meeting. Such preparations will be possible only if individuals are tasked to do them and their home institutions ensure that the time is available for such preparations.

The above conditions for real progress on this Term of Reference are demanding, but for many reasons we feel they are worth the effort. It is well known that models are essential in ecology (like all sciences). They are useful in describing the state of a complex system, and essential in making predictions about its future, or its possible reactions to perturbations. Ecosystem models are essential to the work of WGECO, because of the need to tease out the role of fisheries in observed changes to marine ecosystem components and processes. The changes have multiple causes, and environmental factors have shown time trends while fisheries have increased in effort, changed and adapted gears, and redistributed spatially. Ecosystem models cannot disentangle these causal factors without being combined with good experimentation, but models will be used extensively, and their strengths and weaknesses must be understood better.

The understanding of ecosystem models is needed for more than just evaluating effects of fisheries on ecosystems. Large investments are being made in building capacity for ecosystem management, and ecosystem management requires evaluating the state of marine ecosystems. Models are essential there as well, particularly with regard to evaluating indirect effects of any activity, including fishing, or medium- and long-term magnitudes and consequences of even direct effects. Ecosystem managers and their advisors will never be able to know which metrics of ecosystem status are reliable without knowing the reliability of the models which produced the metric, and basing management decisions on metrics which are unreliable may be extremely costly to both ecosystems and economies.

For all these reasons, ecosystem models contribute increasingly to advisory processes, inside and outside of ICES. There will be immense value in sorting out reliable from unreliable models, informative from misleading indices and metrics, and bounding the conditions where a model’s performance is considered sound. These needs have long been known in models used to assess the status of single species, and ICES has devoted whole workshops to testing models and metrics in those much simpler contexts (ICES, 1995, 1998). The needs are greater and the risks are greater with ecosystem models, because they are used in so many more contexts than just assessing the effects of fishing on ecosystems. Successfully addressing Terms of Reference such as a)i to a)iii in a workshop format as we propose here would contribute widely to ICES science and advisory functions, and to the goals of marine conservation.

3.5 References

Andersen, K.P., and Ursin, E. 1977. A multi-species extension to the Beverton and Holt theory of fishing, with accounts of phosphorus circulation and primary production. Meddelelser Fra Danmarks Fiskeri- og Havundersøgelser. N.S., 7: 319–435.

Baretta, J.W., Ebenhoh, W., and Ruardij, P. 1995. The European regional seas ecosystem model, a complex marine ecosystem model. Netherlands Journal of Sea Research, 33: 233–246.

Basford, D.J., Eleftheriou, A., Davies, I.M., Irion, G., and Soltwedel, T. 1993. The ICES North Sea benthos survey: the sedimentary environment. ICES Journal of Marine Science, 50: 71–80.

Basson, M., and Fogarty, M.J. 1997. Harvesting in discrete-time predator-prey systems. Mathematical Biosciences, 141 (1): 41–74.

Batchelder, H.P., and Williams, R. 1995. Individual-based modelling of the population dynamics of *Mertridia lucens* in the North Atlantic. ICES Journal of Marine Science, 52: 469–482.

Bianchi, G., Gislason, H., Graham, K., Hill, L., Koranteng, K., Manickchand-Heileman, S., Paya, I., Sainsbury, K., Sanchez, F., Jin, X., and Zwanenburg, K. In press. Impact of fishing on demersal fish assemblages. ICES Journal of Marine Science.

Billen, G. 1978. A budget of nitrogen cycling in North Sea sediments off the Belgian coast. Estuarine and Coastal Marine Science, 7: 127–146.

Billen, G., and Lancelot, C. 1988. Modelling benthic nitrogen cycling in temporate coastal ecosystems. *In* Nitrogen Cycling in Coastal Marine Environments, pp. 343–378. Ed. by T.H. Blackburn and J. Sorensen. Wiley and Sons Ltd., London.

Brey, T. 1990. Estimating productivity of macrobenthic invertebrates from biomass and mean individual weight. Meeresforschungen, 32: 329–343.

Bryant, A.D., Heath, M.R., Broekhuizen, N., Ollason, J.G., Gurney, W.S.G., and Greenstreet, S.P.R. 1995. Modelling the predation, growth and population dynamics of fish within a spatially-resolved shelf sea ecosystem model. Netherlands Journal of Sea Research, 33: 407–421.

Carpenter, S.R., Kitchell, J.F., and Hodgson, J.R. 1985. Cascading trophic interactions and lake productivity. BioScience, 35: 634–639.

Cohen, E.B., Grosslein, M.D., Sissenwine, M.P., Steimle, F., and Wright, W.R. 1982. Energy budget of Georges Bank. *In* Multispecies Approaches to Fisheries Management Advice, pp. 95–107. Ed. by M.C. Mercer. Canadian Special Publication in Fisheries and Aquatic Science, 59.

Collie, J.S., and Spencer, P.D. 1994. Modeling predator-prey dynamics in a fluctuating environment. Canadian Journal of Fisheries and Aquatic Sciences, 51: 2665–2672.

Connell, J.H. 1975. Some mechanisms producing structure in natural communities. *In* Ecology and Evolution of Communities, pp. 460–490. Ed. by M.L. Cody and J.M. Diamond. Belknap Harvard University Press, Cambridge, Massachusetts.

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

Death, R.G., and Winterbourn, M.J. 1995. Diversity patterns in stream benthic invertebrate communities: the influence of habitat stability. Ecology, 76: 1446–1460.

Dobby, H., Veitch, A.R., Banks, D., and Gurney, W.S.C. 1999. Incorporating length-based predation into multispecies fisheries modelling. Submitted for publication.

Duineveld, G.C.A., Künitzer, A., Niermann, U., de Wilde, P.A.W.J., and Gray, J.S. 1991. The macrobenthos of the North Sea. Netherlands Journal of Sea Research, 28: 53–65.

Fenchel, T. 1975. Character displacement and coexistance in mud snails (Hydrobiidae). Oecologia, 20: 19–32.

Fransz, H.G., Mommaerts, J.P., and Radach, G. 1991. Ecological modelling of the North Sea. Netherlands Journal of Sea Research, 28: 67–140.

Fransz, H.G., and Verhagen, J.H.G. 1985. Modelling research on the production cycle of phytoplankton in the Southern Bight of the North Sea in relation to river-borne nutrient loads. Netherlands Journal of Sea Research, 19: 241–250.

Furness. R.W. 1978. Energy requirements of seabird communities: a bioenergetics model. Journal of Animal Ecology, 47(1): 39–53.

Gislason, H. 1999. Biological reference limits for interacting species in the Baltic. ICES Journal of Marine Science, 56: 571–583.

Gislason, H., and Lassen, H. 1997. On the linear relationship between fishing effort and the slope of the size spectrum. ICES CM 1997/DD:05.

Heip, C., Basford, D., Craeymeersch, J.A., Dewarumez, J.-M., de Wilde, P., Dorjes, J., Duineveld, G., Eleftheriou, A., Herman, P.M.J., Niermann, U., Kingston, P., Künitzer, A., Rachor, E., Rumohr, H., Soetaert, K., and Soltwedel, T. 1992. Trends in biomass, density and diversity of North Sea macrofauna. ICES Journal of Marine Science, 49: 13–22.

Hollowed, A.B., Bax, N., Beamish, R., Collie, J.S., Fogarty, M., Livingston, P., Pope, J., and Rice, J. In press. Are multispecies models an improvement on single-species models for measuring impacts on marine ecosystems? ICES Journal of Marine Science.

Huston, A.H. 1979. A general hypothesis of species diversity. American Naturalist, 113: 81–101.

Huston, A.H. 1994. Biological diversity: The coexistence of species on changing landscapes. Cambridge University Press. 681 pp.

ICES. 1995. Report of the Working Group on Methods of Fish Stock Assessment. ICES CM 1995/ASSESS:11.

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

Jones, R. 1982. Species interactions in the North Sea. Canadian Special Publication of Fisheries and Aquatic Sciences, 59: 48–63.

Jones, R. 1984. Some observations on energy transfer through the North Sea and Georges Bank food webs. Rapports et Procès-Verbaux des Réunion Conseils International pour l'Exploration de la Mer, 183: 204–217.

Livingston, P.A., and Methot, R. 1998. Incorporation of predation into a population assessment model of eastern Bering Sea walleye pollock. Fishery stock assessment models of the 21st century. Alaska Sea Grant College Program Publication, AK-SG-98-01.

MacArthur, R.H. 1970. Species packing and competitive equilibrium for many species. Theoretical Population Biology, 1: 1–11.

MacArthur, R.H., and Levins, R. 1967. The limiting similarity, convergence and divergence of coexisting species. American Naturalist, 101: 377–385.

MacCall, A.D. 1990. Dynamic geography of marine fish populations. University of Washington Press, Seattle, WA. 153 pp.

May, R.M., Beddington, J.R., Clark, C.W., Holt, S.J., and Laws, R.M. 1979. Management of multispecies fisheries. Science, 405: 267–276.

Menge, B.A., and Sutherland J.P. 1987. Community regulation: variation in disturbance, competition and predation in relation to environmental stress and recruitment. American Naturalist, 130: 730–757.

Mirza, F.B., and Gray, J.S. 1981. The fauna of benthic sediments from the organically enriched Oslofjord, Norway. Journal of Experimental Marine Biology and Ecology, 54: 181–207.

Mommaerts, J.P., Pichot, G., Ozer, J., Adam, Y., and Baeyens, W. 1984. Nitrogen cycling and budget in Belgian coastal waters: North Sea areas with and without river inputs. Rapports et Procès-Verbaux des Réunions Conseil International pour l'Exploration de la Mer, 183: 57–69.

Murphy, E.J., and nine co-authors. 1998. Interannual variability in the South Georgia marine ecosystem: biological and physical sources of variation in the abundance of krill. Fisheries Oceanography, 7: 219–234.

Paine, R.T. 1966. Foodweb complexity and species diversity. American Naturalist, 100: 65–75.

Park, T. 1948. Experimental studies of interspecific competition. I. Competition between populations of the flour beetles *Tribolium confusum* Dival and *Tribolium castaneum* Herbst. Ecological Monographs, 18: 267–307.

Park, T. 1954. Experimental studies of interspecific competition. II. Temperature, humidity and competition in two species of *Tribolium*. Physiological Zoology, 27: 177–238.

Pascual, M., and Levin, S.A. 1999. From individuals to population densities: searching for the intermediate scale of nontrivial determinism. Ecology, 80: 2225–2236.

Patrick, R. 1975. Structure of stream communities. *In* Ecology and Evolution of Communities, pp. 445–459. Ed. by M.L. Cody and J.M. Diamond. Belknap Harvard University Press, Cambridge, Massachusetts.

Pianka, E.R. 1973. The structure of lizard communities. Annual Review of Ecology and Systematics, 4: 53–74.

Pianka, E.R. 1975. Niche relations of desert lizards. *In* Ecology and Evolution of Communities, pp. 292–314. Ed. by M.L. Cody and J.M. Diamond. Belknap Harvard University Press, Cambridge, Massachusetts.

Pianka, E.R. 1983. Evolutionary Biology, 3rd Edition. Harper and Row, New York.

Polovina, J.J. 1984. Model of a coral reef ecosystem I. The ECOPATH model and its application to French Frigate Shoals. Coral Reefs, 3: 1–11.

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

Rosenzweig, M.L. 1971. Paradox of enrichment: destabilization of exploitation ecosystems in ecological time. Science, 171: 385–387.

Rosenzweig, M.L. 1995. Species diversity in time and space. Cambridge University Press, UK. 436 pp.

Rosenzweig, M.L., and Abramsky, Z. 1993. How are diversity and productivity related? *In* Species diversity in ecological communities: historical and geographical perspective, pp. 52–65. Ed. by R. Ricklefs and D. Schluter. University of Chicago Press, Chicago, Illinois.

Ruadij, P., Barretta, J.W., and Barretta-Bekker, J.G. 1995. SESAME, a software environment for simulation and analysis of marine ecosystems. Netherlands Journal of Sea Research, 33: 261–270.

Russo, A.R. 1982. Temporal changes in fish community structure near a sewage ocean outfall, Mokapu, Oahu, Hawaii. Marine Environmental Research, 6: 83–98.

Sainsbury, K.J. 1991. Application of an experimental approach to management of a tropical multispecies fishery with highly uncertain dynamics. ICES Marine Science Symposia, 193: 310–320.

Sanchez, F., and Olaso, I. 1999. Fisheries impacts in the Cantabrian Sea using a mass-balance model. WGECO 1999/10. 19 pp.

Shin, Y.-J., and Cury, P. 1999. OSMOSE: a multispecies individual-based model to explore the functional role of biodiversity in marine ecosystems. Ecosystem considerations in fisheries management. Alaska Sea Grant College Program, AK-SG-99-01.

Sparre, P. 1991. Introduction to multispecies virtual population analysis. ICES Marine Science Symposia, 193: 12–21.

Spencer, P.D., and Collie, J.S. 1996. A simple predator-prey model of exploited marine fish populations incorporating alternative prey. ICES Journal of Marine Science, 53: 615–628.

Steele, J.H. 1974. The structure of marine ecosystems. Blackwell, Oxford. 128 pp.

Steele, J.H., and Henderson, E.W. 1981. A simple plankton model. American Naturalist, 117: 676–691.

Stefànsson, G., and Pàlsson, Ó.K. 1997. BORMICON. A Boreal Migration and Consumption Model. Marine Research Institute Report 58, Reykjavík. 223 pp.

Tjelmeland, S., and Bogstad, B. 1998. Multspec—a review of multispecies modelling project for the Barents Sea. Fisheries Research, 37: 127–142.

Vinter, M., and Thomsen, L. 1998. *Ad hoc* multispecies VPA tuning, an extension to 4M. ICES Study Group on Multispecies Model Implementation in the Baltic. Working Document 1998.

Walters, C.J., Christensen, V., and Pauly, D. 1997. Structuring dynamic models of exploited ecosystems from trophic mass-balance assessments. Reviews in Fish Biology and Fisheries, 7: 139–172.

Walters, C.J., Pauly, D., and Christensen, V. 1998. Ecospace: a prediction of mesoscale spatial patterns in trophic relationships of exploited ecosystems, with emphasis on the impacts of marine protected areas. ICES CM 1998/S:4. 22 pp.

Walters, C.J., Stocker, M., Tyler, A.V., and Westreim, S.J. 1986. Interaction between Pacific cod (*Gadus macrocephalus*) and herring (*Clupea harengus pallasi*) in the Hecate Strait, British Columbia. Canadian Journal of Fisheries and Aquatic Sciences, 43: 830–837.

4 ECOSYSTEM EFFECTS OF FISHING ACTIVITIES IN THE BALTIC SEA

Environmental effects of the Baltic Sea fishery are substantial, in particular at the top of the food web. Cod, which is the dominant piscivorous fish in the Baltic, has been reduced to low biomass by the fishery. Changes in the cod stock have undoubtedly influenced its primary fish prey – sprat and herring. These fishes themselves are dominant zooplanktivores in the ecosystem. Other effects of fishing, on invertebrates, marine mammals, and birds are poorly known.

The ICES Study Group on Ecosystem Effects of Fishing Activities was established in 1990 (ICES C.Res. 1990/2:6) and became a working group in 1992 (ICES C.Res. 1992/2:4). This group has produced several reports on relevant investigations, primarily in the North Sea (ICES, 1994, 1995a, 1996a, 1998). There are, however, distinct differences in the hydrography and bathymetry of the Baltic Sea which are reflected in, e.g., low species diversity. Therefore, there are likely to be important differences between ecosystem effects of fishing activities in the Baltic Sea compared to the effects which ICES has reported for the North Sea.

4.1 Description of the Fisheries

The commercially most important fisheries are for cod, herring, sprat, and salmon.

The main fisheries for cod in the Baltic Sea are those using demersal trawls, high opening trawls (operating both pelagically and demersally), and gillnets. There was an increase in gillnet fisheries in the 1990s and the share of the total catch of cod taken by gillnets has been about 50 % in recent years. Baltic herring is exploited mainly by pelagic trawls, demersal trawls and, during the spawning season, by trap nets and/or pound nets in coastal areas. The main part of the sprat catch is taken by pelagic trawling and used for industrial purposes. Baltic salmon is exploited offshore by drift net and longline while feeding in the sea and with coastal gillnets and traps during the spawning run.

The coastal fishery targets a variety of species with a mixture of gears including fixed gears (e.g., gillnets, pound and trap nets and weirs) and Danish seines. The main species exploited are herring, flounder, turbot, cod, eel, pike, perch, pikeperch, smelt, and whitefish. In addition, there are also demersal trawling activities for herring, cod, and flatfish in coastal waters of the Baltic Sea. Coastal fisheries are conducted along the entire Baltic coastline and in some areas the sport fishery is likely to catch more fish than the commercial fishery. Angling is mainly targeting piscivorous species.

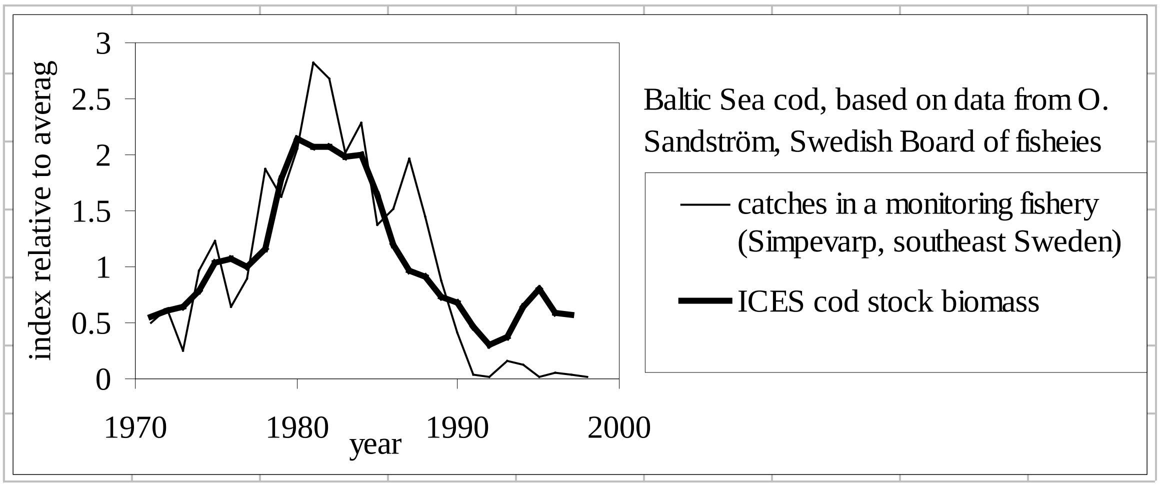
4.2 Effects on Target Species and the Food Web

The main ecological effect of the fishery is the removal of large quantities of fish, in particular target species. Two potential consequences of this are reductions in offspring production and changes in food web dynamics. In the Baltic, there are strong indications that the fishery has caused such changes (Hansson, in press). When cod, the dominant piscivorous fish in the Baltic, had low reproduction success because of hydrographic conditions (Sparholt, 1996; Jarre-Teichman *et al.*, in press), the fishery intensified the decrease and depressed the eastern cod stock to a biomass outside safe biological limits (ICES, 1999a, 1999b). As the cod stock decreased, the predation pressure on herring and sprat was reduced and particularly sprat increased substantially (ICES, 1999b). It has been proposed that the Baltic offshore fish community has gone through a regime shift, from being controlled by cod predation to a clupeid-ontrolled system due to the predation of herring and sprat on eggs and larvae of cod (Köster and Möllmann, 1997)

The growth and condition of herring deteriorated at the same time as the sprat stock increased (Raid and Lankov, 1995). These changes in herring correlate to decreases in the abundance of large zooplankton species, but it is not clear whether this decrease has been caused by increased zooplanktivory or by changes in hydrography (Flinkman *et al.*, 1998).

A consequence of the decreased cod stock is that the species has almost disappeared from some of its normal distribution areas (the Archipelago Sea, the Bothnian Sea, and coastal areas of the Baltic Proper: O. Sandström, Swedish Board of Fisheries, pers. comm.); see Figure 4.2.1.

Figure 4.2.1. Indices of the overall status of the Baltic cod stock (heavy line), and for only the eastern Swedish coast (thin line).



The coastal fishery by anglers and commercial fishermen has probably also influenced ecosystem structures (Hansson *et al.*, 1997). Generally, this impact is more local than that of the offshore fishery, since most of the coastal fish species are relatively sedentary.

In other intensively fished areas and with a variety of species, studies have shown that fishing can induce changes in growth rate, sex ratios, maturity ogives, and genetic composition of fish populations (Jørgensen, 1990; ICES, 1996b; Rice and Gislason, 1996). The exploitation intensity in the Baltic Sea makes such changes likely to have occurred.

4.2.1 Effects due to by-catch and discarding

The total by-catch of fish is unknown for the Baltic Sea fisheries. Reports generally contain few data on this topic, so quantitative estimates cannot be derived. There are, however, some fisheries that have very high proportions of by-catch. In the fishery for vendace roe in the Bothnian Bay, the by-catch is 92 % by weight (O. Sandström, Swedish

Board of Fisheries, pers. comm.). The Study Group on Estimation of the Annual Amount of Discards and Fish Offal in the Baltic Sea (SGDIB) will address this in the spring 2000.

Another type of discarding activity is the return of fish offal to the sea. When this organic matter is degraded, oxygen is consumed and it has been speculated that this can contribute to further reduce oxygen concentrations in bottom waters. As the analysis in Section 5 shows, this is a negligible problem.

4.3 Impact of Fishing Gear on Benthos

All towed demersal fishing gears cause disturbance of the sea bottom surface and, thus, may impact on structures and processes of the sea floor. A review is given in Section 6 and although based mainly on studies in the North Sea, the results are generally relevant also for the Baltic Sea. Effects in the Baltic are, however, expected to be smaller, since beam trawling does not occur and the benthos is generally dominated by smaller organisms that in the North Sea.

Towed gears that penetrate the sediment considerably impact both benthic infauna and epifauna, whereas other types of gears (e.g., gillnet and trap net, Danish seine) may have effects on some epifauna in some circumstances. Local studies are often needed to quantify the impacts of specific fisheries.

There are some specific case studies of the impact of trawl gears on benthos and benthic habitats in the Baltic Sea:

a) nutrient response to trawling activity;
b) there was evidence of sediment disturbance in the Kiel Bight (remobilization of nutrients and increased release of nutrients and organic material were followed by an increase in oxygen consumption (Krost, 1990));
c) biological response of the invertebrate community to trawling activity in the Kiel Bight.

There were obvious biological impacts on thin-shelled bivalves, and starfish suffered heavy damage, whereas little or no damage occurred to the solid-shelled bivalves. However, it was demonstrated that trawling reduced the mean population size of the solid-shelled *Arctica islandica*. An increase in the proportion of damage with increasing body size was found for several species of mussels. Many epibenthic organisms suffered sub-lethal damages. An increase in predatory and scavenging species feeding on dying and dislocated fauna was also documented in this area (Rumohr and Krost, 1991).

4.4 Effects on Seabirds

Fishing nets, in particular set nets, cause considerable mortality of seabirds in the Baltic Sea. Oldén *et al.* (1988) estimated that some 25,000 seabirds, chiefly common guillemots (*Uria aalge*), died between 1982 and 1988 in the set net fishery for cod in the Kattegat. For the Gulf of Gdansk, Stempniewicz (1994) calculated that about 16,000 long-tailed ducks (*Clangula hyemalis*) and velvet scoters (*Melanitta fusca*) are killed annually in the set net fishery for flatfish and cod, representing 10–20 % of the local wintering population of the species. Similarly, up to 17 % of the maximum winter population of eiders (*Somateria mollissima*) and black scoters (*Melanitta nigra*) were estimated to drown in the same fishery type in the Kiel Bight (Kirchhoff, 1982). There are also reports on guillemots and razorbills (*Alca torda*) killed in the drift net fishery for salmon (ICES, 1995b).

An indirect fishing effect that, to our knowledge, remains to be quantified is anglers’ disturbance of nesting birds.

Fishing activities will also affect the seabird community in other ways. These include the effects of discarding of unwanted catch and offal, which then become important sources of food. In the North Sea, several studies indicate that 70–90 % of the offal will be consumed by seabirds (ICES, 1994). This food source is believed to be responsible for increases in seabird populations over time (ICES, 1994, 1996b).

4.5 Effects on Marine Mammals

4.5.1 Harbour porpoises

In Polish waters, there has been an average by-catch of five harbour porpoises per year during the period 1987–1996, mostly in the salmon fishery. Annual by-catches of 3–5 porpoises in bottom-set gillnets and drift nets have been recorded from the Swedish Baltic coast. An estimated 111 porpoises were by-caught in German fisheries in ICES Area

IIIc (the Kiel and Mecklenburg Bays and inner Danish waters) during the period 1987–1995 (Kock and Benke, 1996), and six were by-caught in 1996. These reports suggest that annual by-catches of porpoises are 0.5–0.8 % of the population in the southwestern part of ICES Area IIId (Baltic Sea), and 1.2 % of the population in ICES Area IIIc. Estimates of the harbour porpoise population size are however uncertain, by-catches possibly underestimated, and these by-catches may be unsustainable (ICES, 1997).

4.5.2 Seals

About 200 seals were estimated to have died per year in Estonian waters, predominantly in fyke nets. Grey seals appear to be more vulnerable to being caught than ringed seals and constitute more than 80 % of the by-catches in Estonian waters. In Poland, a total of nine seals, of which a majority was grey seals, was reported being caught in 1996, predominantly in salmon nets. In German parts of the area, one harbour seal was caught by trawl and one in a fyke net in 1995; no cases were reported in 1996. See also ICES (1996a).

Salmon drift nets are also responsible for seal mortalities in Swedish waters. Available Swedish data for the period 1974–1990 show that 29 of 216 (14 %) fishing net mortalities of grey, ringed and harbour seals were caused by salmon drift nets (ICES, 1995b). A survey along the Swedish coast north of Åland showed that a minimum of 250 grey seals were by-caught in 1996. A realistic approximation of the numbers of seals killed in the Swedish coastal fishery is 400 animals (O. Sandström, Swedish Board of Fisheries, pers. comm.). In the Finnish fishery, 338 seals were reported by-caught in the period 1986–1995, of which 113 (33 %) were from the drift net fishery in the open sea. Most of these were grey seals.

In the Bothnian Bay, most of the seals were caught in traps set for salmon and whitefish, while nets set for salmon and whitefish were more important in the Bothnian Sea. In the Baltic Proper, most of the seals were caught in eel traps and turbot nets. Pups of the year made up about half of the total by-catch.

Although recorded data almost certainly underestimate total by-catches, these appear not to constitute a major threat to the seal populations, because their numbers are increasing (Helander and Härkönen, 1997).

4.6 Salmonid Reproduction Disturbances—M74

For 25 years, the Baltic salmon (and possibly also the brown trout) has suffered from a reproduction disturbance referred to as M74. The M74 frequency dropped for some years, but has increased again in 1999 (frequency around 35 %, Lars Karlsson, Swedish Salmon Research Institute, pers. comm.). The results below are from several research projects reported recently (Ambio 1999, issue 1 and TemaNord, report 530) and from a special Theme Session on the M74 Syndrome and Similar Reproductive Disturbances in Marine Animals (Theme Session U) at the 1999 ICES Annual Science Conference.

The frequency of M74 is strongly positively correlated to the SSB of sprat in the Baltic Sea, and it has been reported that it is caused by salmon gorging on sprat (MacKenzie, 1999). This is however not the explanation, since the proportion of sprat in the diet was the same before M74 was recorded (1959–1962) as during a period with a high frequency of the syndrome (1994–1997). Since M74 can be cured by thiamine, it is likely that the syndrome is caused by a deficiency of this essential vitamin. It has been concluded that this deficiency is most probably caused by changes in the food web, since a similar disease syndrome has been recorded in landlocked salmonids during drastic food web changes in the North American Great Lakes. The decline in the Baltic cod stock, and resulting ecosystem changes, may have caused or contributed to M74. The ultimate cause of M74 is however still unknown, and further research is needed to shed light on the role of the fishery and other environmental changes (e.g., eutrophication and climate).

4.7 Preventive Measures

The key preventive measure to be taken to reduce the ecosystem effects of fishing in the Baltic Sea is to substantially decrease the fishing pressure on cod, allowing the stock to build up. Given the very small population size of the harbour porpoise, urgent measures are needed to decrease the by-catches of this species. Since the frequency of M74 is still high, it is important that the salmon fishery is managed in such a way that the mortality of naturally reproducing populations is sustainable. The ecological impact of various fishing gears is under investigation as well as ways to reduce adverse effects on fish, invertebrates, marine mammals, and seabirds.

4.8 Summary

Fisheries management in the Baltic Sea faces unique challenges. The low and variable salinity in the Baltic Sea has resulted in an ecosystem that has relatively low species diversity and a comparatively simple food web, resulting in an ecosystem of low redundancy (cf. Section 6.3.2). To become sustainable, the fishery has to be managed within these boundary conditions. The fishery has failed to meet these requirements and has contributed substantially to large ecosystem changes. The primary impact of the fishery has thus been the direct removal of target species. Secondary effects are impacts of bottom trawling and by-catches, in particular of harbour porpoises.

4.9 Need for Further Research or Additional Data

Large-scale food web changes have occurred in the Baltic Sea, resulting from fishing and natural variation in environmental conditions combined with effects of eutrophication and other pollutants. These changes, and the great value that the Baltic has to the approximately 85 million inhabitants of the 14 countries in the Baltic Sea drainage area, highlight the need for broad management approaches. Management must increase the focus on the ecosystem effects of the fishery. Since the Baltic Sea is one of the most intensively studied areas of the world's ocean, experiences from this broader management approach will probably be of interest elsewhere.

4.10 References

Flinkman, J., Aro, E., Vuorinen I., and Viitasalo, M. 1998. Changes in northern Baltic zooplankton and herring nutrition from 1980s to 1990s: top-down and bottom-up processes at work. Marine Ecology Progress Series, 165: 127–136.

Hansson, S. In press. Human effects on the Baltic Sea ecosystem—fishing and eutrophication. *In* Ecosystem Considerations in Fisheries Management. Proceedings of the International Symposium on Ecosystem Considerations in Fisheries Management. Alaska Sea Grant Program report, University of Alaska, Fairbanks.

Hansson, S., Arrhenius, F., and Nellbring, S. 1997. Food web interactions in a Baltic Sea coastal area. *In* Forage Fishes in Marine Ecosystems. Proceedings of the International Symposium on the Role of Forage Fish in Marine Ecosystems. Alaska Sea Grant Program report: 97-01 pp. 281–291. University of Alaska Fairbanks, Fairbanks.

Helander, B., and Härkönen, T. 1997. Marina toppkonsumenter. Östersjö, 96: 45–48.

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

ICES. 1995a. Report of the Study Group on Ecosystem Effects of Fishing Activities. ICES Cooperative Research Report, 200. 120 pp.

ICES. 1995b. Report of the Baltic Salmon and Trout Assessment Working Group. ICES CM 1995/Assess:16.

ICES. 1996a. Report of the ICES Advisory Committee on the Marine Environment, 1996. ICES Cooperative Research Report, 217: 68–73.

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

ICES. 1997. Report of the ICES Advisory Committee on the Marine Environment, 1997. ICES Cooperative Research Report, 222: 78–80 and 93.

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

ICES. 1999a. Report of the Baltic Fisheries Assessment Working Group. ICES CM 1999/ACFM:15.

ICES 1999b. Report of the ICES Advisory Committee on Fisheries Management, 1998. ICES Cooperative Research Report, 229.

Jarre-Teichman, A., Wieland, K., MacKenzie, R.R., Aro, E., and Plikshs, M. In press. Stock-recruitment relationships for cod (*Gadus morhua*) in the central Baltic Sea incorporating environmental variability. ICES Journal of Marine Science.

Jørgensen, T. 1990. Long-term changes in age at sexual maturity of northeast Arctic cod (*Gadus morhua* L.). ICES Journal of Marine Science, 46: 235–248.

Kirchhoff, K. 1982. Wasservogelverluste durch die Fischerei an der Schleswig-Holsteinichen Ostseeküste. Die Vogelwelt, 103: 81–89.

Kock, K.-H., and Benke, H. 1996. On the by-catch of harbour porpoise (*Phocoena phocoena*) in German fisheries in the Baltic and the North Sea. Archive of Fishery and Marine Research, 44: 95–114.

Krost, P. 1990. Der Einfluss der Grundschleppnetzfischerei auf Nahrsalz-Freisetzung aus dem Sediment und Makrofauna der Kieler Bucht (Westl. Ostsee). Berichte aus dem Institut für Meereskunde an der Christian-Albrecht Universität Kiel, Nr. 200.

Köster, F.W., and Möllmann, C. 1997. Predation by sprat and herring on early life stages of cod and sprat in the central Baltic. *In* Forage Fishes in Marine Ecosystems. Proceedings of the International Symposium on the Role of Forage Fish in Marine Ecosystems. Alaska Sea Grant Program report: 97-01 pp. 41–69. University of Alaska Fairbanks, Fairbanks, Alaska.

MacKenzie, D. 1999. Junk food. When salmon gorge on sprats, the offspring sicken and die. New Scientist, 162: 2183.

Oldén, B., Peterz, M., and Kollberg, M. 1988. Sjöfågeldöd i fisknät i nordvästra Skåne. Naturvårdsverket Rapport 3414. 51 pp. Solna, Sweden.

Raid, T., and Lankov, A. 1995. Recent changes in the growth and feeding of Baltic herring and sprat in the northeastern Baltic Sea. Proceedings of the Estonian Academy of Sciences, Biology and Ecology, 5: 38–55.

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

Rumohr, H., and Krost, P. 1991. Experimental evidence of damage to benthos by bottom trawling with special reference to *Arctica islandica.* Meeresforschungen, 33: 340–345.

Sparholt, H. 1996. Causal correlation between recruitment and spawning stock size of central Baltic cod? ICES Journal of Marine Science, 53: 771–779.

Stempniewicz, L. 1994. Marine birds drowning in fishing nets in the Gulf of Gdansk (southern Baltic): numbers, species composition, age and sex structure. Ornis Svecica, 4: 123–132.

5 REVIEW PAST INFORMATION ON SECONDARY EFFECTS CAUSED BY DUMPING FISH REMNANTS IN THE BALTIC

In the Terms of Reference, WGECO was asked to evaluate possible effects of offal discards in the Baltic. This issue was raised by HELCOM (1999/5) and in the 1997 ACME Report (Section 12). One aspect addressed was the question of the effects of offal on the oxygen deficiency in bottom waters. To analyse this, we concentrated on the Baltic Proper where most of the fishery takes place, in particular the cod fishery where most fish offal is generated. This area is also where the problem of deep-water oxygen deficiency is most pronounced.

A simple carbon budget (below) clearly shows that the oxygen requirement from discarded offal is insignificant. The oxygen consumption by discarded offal is << 1 % of the consumption by sedimented phytoplankton.

Baltic Proper primary production[1]	170	Elmgren, 1984[3]
sedimentation[1]	57	Elmgren, 1984[3]
discarded offal[1]	0.028	high values from the early 1980s[2]: 58 000 tonnes carbon as proportion of wet weight[3]: 0.1 surface area of the Baltic Proper[3]: 2.1×10^{11} m^2
contribution by offal to total oxygen consumption	**0.05 %**	

[1] g C m^{-2} on an annual basis [2] 1997 ACME Report, p. 92 [3] Elmgren, R. 1984. Trophic dynamics in the enclosed, brackish Baltic Sea. Rapports et Procès-Verbaux des Réunions du Conseil International pour l'Exploration de la Mer, 183: 152–169.

Lack of relevant data prevented us from analysing other possible impacts from by-catch and offal discards. Experiences from the North Sea however shows that seabird community structures may be influenced (1997 WGECO Report, p. 226). By-catches in the Baltic will be analysed during 2000 by the Study Group on Estimation of the Annual Amount of Discards and Fish Offal in the Baltic Sea (SGDIB).

6 BOTTOM TRAWL IMPACTS ON THE BENTHOS IN THE NORTH SEA AND IRISH SEA

6.1 Approach to the Term of Reference

This Term of Reference occupied the majority of time of WGECO participants, both in plenary or in subgroups and individually. WGECO discussed several times how to approach the presentation of the results of our deliberations. WGECO acknowledged the need to pay due heed to the specific Terms of Reference. At the same time it was agreed that we should not act in naive ignorance of the much wider interest in the question of how bottom trawling affects ecosystems, particular their benthic components, and the corresponding likelihood that our conclusions would be interpreted more widely that just within the context of benthic taxa in the North Sea and Irish Sea.

In accordance with the IMPACT II Report, the phrase ‘bottom trawls’ refers to beam trawls and otter trawls, which have been used in recent years in the southern North Sea and in the Irish Sea. However, WGECO acknowledges that many other bottom gears are in use in various fisheries in the ICES area and beyond, and that even beam trawls and otter trawls used in the North Sea and the Irish Sea are classes of gears that often undergo technological changes. In the few cases in this section where our conclusions extend beyond beam trawls and otter trawls, we make that extension explicit. Otherwise our conclusions should be restricted to those two gears, as used in the areas considered in the IMPACT II Report.

To ‘consider the report’ of IMPACT II, we chose to evaluate the specific conclusions drawn by the contributors to it. Most of these are summarized in their ‘Summary and Conclusions’ section (Section 5 of the IMPACT II report; pp. 368–371), but occasionally statements which are clearly conclusions are presented in Section 3 of their report but not carried over to Section 5. Our evaluation consisted of first reviewing the research results summarized or reviewed earlier in the volume, with regard to scientific criteria such as soundness of research design, validity of statistical analyses, etc. We often augmented the information base on which we evaluated each conclusion, by also considering publications of additional studies, often from outside the area addressed in IMPACT II, or reported after IMPACT II was completed. Following these reviews of the soundness of the background information, we present our conclusions on the validity of each conclusion from IMPACT II (which is quoted directly in *italics*). We based our conclusions on the extent to which each is supported by the results of the studies comprising IMPACT II, the soundness of those studies, and the degree of support from the other studies reviewed. All this information is contained in Section 6.2 below, and summarized in Table 6.2.5.1.

As these conclusions were evaluated, WGECO discussed several times exactly what ‘Effects’ could be attributed to bottom trawling, and what classes of measures could be implemented to ‘Mitigate’ the effects. Possible effects were found to fall into two categories—effects on ‘Habitats’ and effects on ‘Species’. Mitigative measures also fell into two classes—those with an inherent spatial character and those without an intrinsic spatial aspect. Each time a sub-section of Section 6.2 provides results relevant to our list of effects, the support (or lack thereof) for the effect is pointed out explicitly. In Sections 6.3.1 and 6.4, respectively, we summarize each candidate effect and each mitigative measure concisely, and relate measures to which effects they could mitigate (Table 6.4.1). We conclude that treatment with an evaluation of which candidate effects actually are documented to have resulted from bottom trawling in the North Sea and Irish Sea, and therefore measures to reduce or eliminate them might be warranted. We stress that 'no evidence' cannot be interpreted as confidence that the effect does not occur. Rather, WGECO saw no scientifically credible evidence that the effect has been demonstrated, at least for benthos, but remains open-minded to the possibility that new information may be brought forward in the future. WGECO also notes that many management measures implemented to reduce effects for which evidence was found are likely to have conservation benefits for the undocumented effects, if those effects were to be occurring.

Effects on communities and ecosystems were also discussed, but it was realized early in the meeting that these effects would be less direct that the effects addressed in Section 6.3.1. Consistent with Section 8 of our previous report (ICES, 1998), and Section 7 of this report, we agreed that evidence for such higher-order effects would probably be more elusive and subject to controversy, and potential mitigative measures might not be apparent to all. Nonetheless, we agreed that it was important to address this important topic, and it is treated separately in Section 6.3.2.

Section 6.5 integrates the preceding material. It presents WGECO's evaluation of which effects are of sufficient concern to warrant measures be implemented to reduce them, and the measures that WGECO considers most appropriate for pursuing each desired reduction. The likelihood that the recommended measures may contribute to reducing other direct and indirect ecosystem effects of fishing (if they are, in fact, occurring but are not detected) are also noted. The section also comments on WGECO's evaluation of the extent to which each measure may reduce the opportunity to catch commercially important fish.

6.2 Background Information from IMPACT II and the Scientific Literature

6.2.1 Fleets, gears, and physical impacts

6.2.1.1 Introductory remarks on gear types used in the report and fishing effort

When drawing conclusions from the IMPACT II report, it is essential to take account of the limited number of gear types that have been used in the reported experiments. Beam trawls with chain mats or tickler chains used in the Dutch fishery and a *Nephrops* trawl used in the Irish Sea are the only commercial fishing gears studied. The IMPACT II conclusions do not fully take account of the fact that within each gear type there are several possible designs of the individual components. For example, otterboards which are towed by hinged brackets produce far less ground shear than some other designs; fishermen may increase the diameter of groundgear and wires by a factor typically between 2 to 5 in order to prevent them from digging into soft substrates. In each of these cases, the extremes of design will demonstrate quite different effects on any given type of seabed and its associated benthic communities.

There is an increasing trend, not acknowledged in the report, that fishing gears are being designed and operated specifically for particular fisheries. It may not be appropriate to extrapolate the effects found in the report to other fisheries using other gear designs on different habitats.

The impact of fishing is implicitly assumed to bear some relation to the effort expended, which in turn is assumed to relate to catch of target species in a fixed way. However, there are several reasons why effort and catch do not have a constant relationship over time, across gear types, or among countries:

1) Effort is strictly the product of fishing activity and fishing power. Often fishing activity, such as hours fishing, is used as a nominal representation of fishing effort, but it may ignore the difference in catching power between gears.
2) The rapid development of commercial fishing gear design in the past twenty years is continuing. A vessel of a given horsepower may tow a single trawl, a twin or multiple trawl, a scraper trawl targeting fish close to the ground or, in partnership with a second vessel, a pair trawl. Each gear may be targeting a different mix of species and each may have a markedly different fishing power for each species. Sangster and Breen (1998) compare the fishing power of single and twin trawls.
3) The convention for recording fishing activity varies among countries so that one hour's fishing with a beam trawl has a different meaning in Belgium and the UK, for example.
4) Generally the analysis of fishing effort is done by grouping vessels within particular bands of registered engine horsepower, recognising that horsepower is related to gear size. However, the useful proportion of this nominal engine power, i.e., that used for towing the gear, has increased significantly over recent years. Propeller nozzles have increased efficiency and propulsion systems are now designed such that auxiliary equipment is no longer

driven from the main engine. Increases of up to 100 % are plausible (English, 1993). This allows a vessel to tow a bigger gear or a similar-sized gear at a higher speed.

6.2.1.2 Size of bottom trawling fleets

By the beginning of the Twentieth Century, the North Sea was already intensively fished. Due to very scarce historical data on fishing vessels, it is difficult to clearly assess the historical impact of the fishery on the environment. WGECO attempted to develop a measure of effort for the North Sea in earlier meetings (ICES, 1993), and no additional information has become available to alter the earlier estimates of historic effort trends. With regard to effort in recent years, information on numbers of vessels and landings is available in IMPACT II only for the Belgian, Dutch and German fleets. Fleets from several other countries, particularly the UK, Ireland, France, Denmark, and Norway, also operate regularly in various parts of the North Sea and Irish Sea, and a full description of the types of gears used in those seas, and trends in effort, would have to include data on those fleets as well.

Conclusions from the IMPACT II report

'Fishing has been an important industry since the beginning of this century. … The fishing industry at that time used mainly passive fishing gears but trawl nets were already in use by steam trawlers and larger sailing vessels. Technological advances have led to an increase in the impact, with the introduction of the diesel engine, otter trawls able to fish rough grounds, the beam trawl and modern navigation equipment as the main steps.' (IMPACT II, p. 368).

WGECO concurs that there is no doubt that the fishing effort applied in many fisheries has increased greatly in the historic period, and is steadily increasing, even though the fishing activity may remain relatively constant.

6.2.1.3 Fishing gears used by different fishing fleets

A vessel and gear inventory of the Belgian, Dutch and German fishing fleets operating in the North Sea was made to obtain the technical and operational details categorised by engine power (kW).

Integrated effort data (hours fished) from these fleet categories were only available for 1994 and were presented on and ICES-rectangle basis. From these data, it is not possible to indicate whether specific areas within a rectangle have been fished or not.

Conclusions from the IMPACT II report

'*Beam trawls are the most common demersal fishing gears in Belgium, The Netherlands and Germany at present. Otter trawling has a minor importance and its use is still decreasing. For the UK and Ireland the otter trawl is the most frequently used fishing gear. In the U.K. however, beam trawling has gained an increasing importance over recent years*' (IMPACT II, p. 368).

The report does not address the ways in which fishermen vary their gear types and/or methods of deployment. This process of change can occur very quickly in some circumstances and, as a result, will alter the profile of impacts from the fleet sector concerned. As an example of this phenomenon, there is currently a pronounced change from the use of beam trawling to multi-rig otter trawls. This can be observed in the Netherlands, Belgium and England. It is now the case that beam trawl effort is declining in some areas and a form of otter trawling is increasing, contrary to the above conclusion (Anon., 1999).

Additional information from other studies:

1) Information on the fleets structure operating in the North Sea is available in the so-called 'Lassen-Report' (Anon., 1996), which was prepared by a group of independent experts to advise the European Commission on the Fourth Generation of the Multi-annual Guidance Programmes in 1996.
2) The microdistribution of fishing effort of the Dutch beam trawl fleet (13 % of the fleet fishing with 12-m beam trawls) is shown by Rijnsdorp *et al*. (1998).
3) For the period 1990 to 1995, the distribution of international effort (hours fished) in the North Sea on a rectangle basis for beam and otter trawlers is presented by Jennings *et al*. (1999).

4) Although the report suggests that the importance of otter trawling has decreased for fleets of the three countries whose data are summarised in the IMPACT II report, otter trawls are still widely used by fleets based in Norway, Denmark, Scotland, England and Ireland.

Although the major direct impacts of fishing on benthic communities in these waters are likely to be caused by beam trawls and otter trawls, impacts on many benthic populations should be considered in light of all the fisheries, and not just the beam trawl and otter trawl fleets.

6.2.1.4 Physical impacts

The pressure exerted by the sole plates of a beam trawl was directly measured during experiments at sea with 4-m and 12-m beams. The seabed disturbance was studied for 4-m and 12-m beam trawls rigged with tickler chains, for a 4-m beam trawl rigged with a chain matrix, for a *Nephrops* otter trawl, and for a rockhopper ground gear rig attached to otterboards but with no net.

The conclusions from the IMPACT II report relate mainly to the beam trawl experiments, and are scattered through both Sections 3 and 5. These are that:

- the pressure force exerted on the sea floor is strongly related to towing speed;
- the greater weight of the 12-m beam trawl is compensated by the larger sole plate dimensions and higher towing speed so that the pressure exerted by a 12-m beam is more or less the same as that for a 4-m beam;
- the penetration depth of a beam trawl depends, *inter ali*a, on sediment characteristics and varies between 1 cm and 8 cm;
- heavy beam trawls disturb at least the upper 1 cm of the sediment;
- on fine and very fine sand, the bottom becomes harder and less rough after the passage of the gear but in less than 15 hours the original situation is restored.

In addition and applying to both beam and otter trawls:

- the movement of the gears over the seabed causes the suspension of the lighter sediment fractions;
- the sole plates of beam trawls and the boards of otter trawls leave detectable marks on the sea floor;
- depending on local circumstances, these marks disappear after a period which depends on sediment and tidal conditions. Examples given in the study are 37 hours for beam trawl tracks in conditions of fine sediments and tidal currents, to 18 months in muddy sediments and sheltered conditions.

Where the short- and long-term physical effects are concerned, the experimental methods devalue the data in two key respects:

- the use of different sea areas for each study precludes comparisons between the two sets of data;
- the experimental rig in Loch Gareloch was completely unlike commercial fishing gear because no net was attached to the ground gear. This would have affected the behaviour of the gear quite significantly. One example of this is the dynamic equilibrium of forces acting on the otter boards and the ground gear. In the circumstances of the experiment, the ground shearing forces from the boards and the digging action of the ground gear would be atypical. The physical impacts exerted by these components were therefore not necessarily representative. If this is the case, then it would affect any conclusions drawn as to the relative depths to which beam and otter trawls disturb the seabed sediments.

Additional information from other studies:

1) concerning the penetration depth of a 12-m beam trawl, the preliminary results from the EU-project ‘TRAPESE’ (Trawl penetration in sediments) support the findings presented in the IMPACT II report (M. Bergman, pers. comm.);
2) beam trawl fishing, even in areas of very high fishing effort, is highly patchy (Rijnsdorp *et al.*, 1999). Over a period of 6 years (1993–1998), about 20 % of the southeastern North Sea (roundfish area 6) was estimated to be trawled at an intensity below 1 time per 5 years. Other areas however are shown to be trawled at high densities of more than 10 times per year. Estimations of trawling intensities related to the entire North Sea (Ehrich, 1998) show greater spatial differences. Only 2 % of a lightly fished ICES rectangle is affected by bottom trawls. Based

on swept area calculations, the sediments in heavily fished areas are disturbed four times per year. Taking the behaviour of fishermen and the highly patchy distribution of effort into account, Ehrich came to the same maximal estimate of more than 10 times per year for an area being trawled.

6.2.1.5 Evidence in support of the effects of bottom trawls

Sections 3.1 and 3.2 of the IMPACT II report provide no evidence for effects on habitat, nor do Sections 3.1 to 3.3 address possible effects on species or populations.

Effects on Habitats

A more detailed consideration of effects on habitats is given in Section 6.3, below.

Bottom trawling can remove some physical features

There is no evidence for this effect in the report. Reference to other literature that supports this conclusion is provided in Sections 6.2.3.6.1 and 6.2.4.2.1.

Bottom trawling can cause a reduction in structural biota (biogenic features)

There is no evidence for this effect in the report. However, evidence is provided in the broader literature that is referenced in Section 6.2.4.2.1 of this report. Further evidence is provided by Schwinghamer *et al.* (1998), who noted the destruction of biogenic structures such as tubes and burrows and the loss of flocculated organic matter on the sediment surface in response to otter trawling at a deep-water sandy bottom site on the Grand Bank. Recovery from disturbance was in the order of one year in the absence of repeated trawling.

Bottom trawling can cause a reduction in complexity

The findings presented in the IMPACT II report also support this effect of bottom trawling. In addition, Schwinghamer *et al.* (1998) noted changes in the acoustical properties of the upper 4.5 cm of sediment with otter trawling at a deep-water sandy bottom site on the Grand Bank, which were interpreted as a decrease in habitat complexity. However, this conclusion was confounded by the loss of structural biota noted above.

Bottom trawling can cause a reduction in the physical structure of the sea floor

The findings presented in the IMPACT II report also support this effect of bottom trawling. There is support elsewhere for the opposite conclusion in the short term. Schwinghamer *et al.* (1998) found an immediate effect of otter trawling at a sandy bottom site on the Grand Bank (approximately 130 m depth). The trawl doors created berms and furrows that were seen by sidescan sonar and also with video imagery. The persistence of these features was in most cases less than one year in the absence of further trawling.

6.2.1.6 References

Anon. 1996. Report of a Group of Experts to advise the Commission on the Fourth Generation of the Multiannual Guidance Programme. EC DG XIV. April 1996.

Anon. 1999. ‘Beam trawlers switch to twin-rigging’, p. 3. Fishing News, 27 August 1999.

Ehrich, S. 1998. Entwicklung und geographische Verteilung des Fischereiaufwandes in der Nordsee. German Journal of Hydrography, Suppl. 8: 85–89.

English, H.R. 1993. Towing power efficiency. Seafish Industry Authority Report, 417. 20 pp.

ICES. 1992. Report of the Study Group on Ecosystem Effects of Fishing Activities, Copenhagen, 7–14 April 1992. ICES CM 1992/G:11.

ICES. 1998. Working Group on Ecosystem Effects of Fishing Activities. ICES CM 1998/Assess/Env:1.

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

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

Rijnsdorp, A.D., and Piet, G.J. 1999: Effect of beam trawl fisheries on the North Sea benthic ecosystem. WGECO 1999/8. 7 pp.

Sangster, G.I., and Breen, M. 1998. Gear performance and catch comparison trials between a single trawl and a twin-rigged gear. Fisheries Research, 36: 15–26.

Schwinghamer, P., Gordon, D.C., Rowell, T., Prena, J., McKeown, D.L., Sonnichsen, G., and Guigné, J.Y. 1998. Effects of experimental otter trawling on surficial sediment properties of a sandy bottom ecosystem on the Grand Banks of Newfoundland. Conservation Biology, 12: 1215–1222.

6.2.2 Direct mortality

6.2.2.1 Catch efficiency of commercial trawls

Section 3.4 of IMPACT II reports on a series of field studies in which benthic fauna were sampled using grabs, dredges and box corers. Subsequently, the sites were fished using commercial trawls and then resampled. Studies were conducted during 1992–1995 in the southern North Sea on sandy and silty substrates using 4-m beam trawls, 12-m beam trawls and otter trawls, and in the Irish Sea during 1994–1996 on muddy substrate using a *Nephrops* trawl. The study produced estimates of catch composition for these gear types (including both retained and discarded portions). The study also sought to determine the catch efficiency of the gear by size category for different invertebrate and small fish species, defined as the proportion of the individuals in the path of the gear that are captured by the commercial gear.

6.2.2.1.1 Catch composition

Assorted research fishing was used to examine the composition of catches by different gears in various areas. Although this work was intended to mimic commercial fishing, estimates of by-catch ratios and other characteristics in research catches cannot be treated as equivalent to values recorded aboard commercial vessels engaged in their normal operations. Such parameters are highly sensitive to the practices by which fishermen target their gear onto aggregations of the desired species. Furthermore, as the authors state, the catch composition varied with geographical area as a function of the relative distribution of the species.

‘In North Sea flatfish beam and otter trawls, the by-catch by weight of flatfish (mostly dominated by dab and plaice) was at least as high as the marketable catch. The by-catch of invertebrates (dominated by starfish, heart urchins, crabs) was several times the amount of marketable fish. The by-catch weight of roundfish was relatively low (< 5 % of the total catch).’ (IMPACT II, p. 369).

We treat this ‘conclusion’ as a summary of results, and note that the observed catch composition in this study is a result of the trawl’s catching efficiency and the initial density and size/species distribution. The results hold through a range of catch rates (from mean catch rates less than 10 kg/ha trawled up to more than 100 kg/ha), but only for beam trawls. The samples are taken from a range of locations with both sandy and silty bottom conditions and covering several seasons and areas, but are not representative of the North Sea beam trawl fisheries. Also, otter trawls are not very well represented and the results of this study may not be applicable to areas in the northern part of the North Sea where that fleet concentrates. It should also be emphasized more that on only a few occasions did the marketable fish constitute more than 20 % of the total catch.

‘In the Irish Sea Nephrops studies, the by-catch by weight of roundfish (mainly juvenile whiting) was roughly similar to the amount of marketable prawns. The by-catch of non-target invertebrates was relatively low (< 5 % of the total catch) and dominated by crustaceans and molluscs. However, the recent implementation of legislation governing the insertion

of square mesh panels in Nephrops nets will increase the quantity of fish escaping from the net, and thus reduce the quantities of juvenile whiting discarded.' (IMPACT II, p. 369).

The first two sentences of this conclusion are only a summary of results and these are well supported in the report. It is quite reasonable to believe that the insertion of square mesh panels will increase the escapement of fish from the trawl, although direct evidence for this is not provided in the report.

'The catch efficiency for invertebrates did not differ between 4 m and 12 m beam trawls both rigged with tickler chains. In 4 m beam trawls with tickler chains, more marketable fish and invertebrates (by weight) were caught that in 4 m beam trawls rigged with a chain matrix. The total catch by weight in 12 m beam trawls was several times higher than in otter trawls: for marketable fish at least seven times, for all discards more than ten times.' (IMPACT II, p. 369).

Much of this statement is an iteration of results, and not a general conclusion. The report discusses the results of various analyses of variance (ANOVA) which purport, in the headings, to be analyses of catch composition. In fact, the analyses performed were of catch rate and not catch composition. From an analytical point of view, the conclusions of the ANOVAs presented in this section (IMPACT II, pp. 164–165) do not take into account the number of tests performed (65 in this one study). When the critical probability is Bonferroni adjusted (i.e., $P = 0.008$), the 'significant' differences between the June samples collected from a 4-m beam trawl with and without the chain matrix become non-significant for all but marketable sole. Similarly, the differences between the 12-m and 4-m beam trawls disappear at that time of the year as well. The only remaining differences occur between the beam and otter trawl gear (the latter catching fewer fish or invertebrates) in the German Bight (but only 5 of 8 significant results stand in the September 1994 experiments), and between the three beam trawl gear types off the Dutch coast in September. It is not clear why these latter results should show significant differences in the catch between the gear types in September but not in July. Water temperature is not an issue when these months are compared, and these differences could reflect spatial heterogeneity.

This section is very descriptive in nature and does not contribute significantly to the conclusions in terms of impact of bottom trawls to benthic communities. It essentially addresses direct removals of target species and by-catch.

6.2.2.1.2 Catch efficiency for small-sized fish and invertebrates

'The catch efficiency of beam trawls was generally low (0 % to 10 %) for small sized fish and invertebrate species. In otter trawls, the catch efficiencies for invertebrates were even lower than 3 %. Only in beam trawls the largest length classes of starfish, hermits and seamouse were caught with efficiency higher than 10 %.' (IMPACT II, p. 369).

This statement is not a conclusion, but instead an iteration of results. The initial density of small fish was estimated using a 3-m beam trawl with fine meshes. An unknown fraction of small-sized fish will escape out of the path and not enter the trawl. This point only strengthens the findings for small-sized fish catch efficiency. It is not obvious from the IMPACT II report how many samples with Triple-D or box corer were taken to estimate the initial density of invertebrates to be compared with the commercial trawl. With a patchy distribution (of at least some species), the estimated efficiency will be given with a higher variance. The overall interpretation of the results is that the commercial trawls are quite size-selective on what is retained in the codend after entering the trawl. This conclusion is not contentious.

It is well documented that the degree to which species are herded by the bridles into the net and the variability in the spread of the gear are two factors critical to the calculation of catch efficiency. Both the herding effect and catch efficiency are known to be size- and species-specific. Further, identical trawls deployed on different boats can produce 25 % variation in wing spread (Fréchet, 1996, 1997). In this report, an otter trawl was markedly less efficient than the beam trawls but that was probably because the swept area was calculated from the door spread, not the wing spread. Electronic measurements (e.g., SCANMAR) of spread would have reduced the variance in this variable. Further, the fish species were evaluated using the door spread while the invertebrates were evaluated using the wing spread to calculate swept area. This was predicated on the reasonable assumption (not proven) that infauna will not be guided along the doors, then along the bridles, and finally into the net itself.

No analyses were performed with these data and inferences refer only to trends in mean values. Different initial densities and size spectra confound all of these data. Although the data were standardized as percentages of the initial densities on the seabed, they do not describe the differences in initial density between the sites that will influence the results.

6.2.2.2 Direct mortality due to trawling

Section 3.5 of the IMPACT II report describes the various types of mortality that occur over the whole fishing process; four types of trawls were used. The occurrence of damage due to contact with tickler chains in a beam trawl was described, and detailed field studies in the North Sea and the Irish Sea were undertaken to calculate the total direct mortality of invertebrates and flatfishes. Estimates of both the mortality of caught animals (including discard mortality) and of the animals in the trawl path (post-selection mortality) were made.

6.2.2.2.1 Mortality of discards

These investigations were intended to estimate direct mortality of non-targeted epibenthic invertebrates and undersized flatfishes caught in the North Sea commercial fisheries. Ship-board experimentally-derived mortality was examined for species caught in 12-m and 4-m trawls with tickler chains, 4-m beam trawls with chain matrix and otter trawls. Percent mortality of animals brought dead aboard (immediate discard mortality), and the percent of animals that died over several hours to three-day periods (secondary discard mortality) were estimated.

‘Mortality of discards from flatfish beam trawls was species-dependent and varied for invertebrates from <10 % of the individuals caught (starfish, brittlestars) to almost 90 % (the bivalve Artica islandica), with most crustaceans showing intermediate values (about 50 % to 70 %). Discarded fish showed mortalities ranging from 50 % to 100 % (flatfish), from 80 % to 100 % (roundfish), with 100 % mortalities for gadoids.’ (IMPACT II, p. 369).

‘For the majority of fish and invertebrate species, no clear differences were found in discard mortalities between the different trawls tested.’ (IMPACT II, p. 369).

‘Despite the high mortality of discarded small fish and most invertebrate species, this mortality is still very low (a few percentage) when expressed as percentage of the initial density of these animals on the seabed. This is due to the low catch efficiency of the commercial trawl for these species which mostly pass through the meshes or do not even enter the net. For all invertebrate species, direct mortality mainly occurred in the trawl path, possibly as a result of direct physical damage inflicted by the passage of the trawl or indirectly from disturbance and subsequent predation.’ (IMPACT II, p. 369).

This portion of the research is not presented with enough detail to evaluate its reliability, and few of the data are analysed statistically to support the results. The statements above summarize the results of the study and draw no conclusions of broader interest. The authors state in the text that no significant differences in discard mortality were observed among the four different trawls. The authors also note that these gears are not very efficient in catching small fish and invertebrates; most of these pass through the mesh or do not even enter the net. These results justify the examination of the post-selection mortality.

The authors confess their inability to demonstrate consistent differences between direct (primary) and indirect (secondary) discard mortality, likely the result of large variation among replicates as well as low sample sizes. As a result, the effects of accumulated catch on potential mortality of discards may be estimated poorly. Also, no post-selection studies of fish species were undertaken, so the overall mortality estimates may be underestimated; post-selection mortality is known to be significant for roundfish off the west coast of Scotland (Anon., 1999) and walleye pollock in Alaska (Erickson *et al*., 1999).

This section contributes significantly to the conclusion section and is consistent with other studies done on discard mortalities.

6.2.2.2.2 Damage of invertebrates

This study investigated the damage to benthic invertebrates due to contact with tickler chains in sandy bottom habitats in the North Sea, using ‘Kieler Kinderwagen’ dredges attached to a 7-m beam trawl. The authors indicate that ‘several’ replicate tows were made, each haul covering a distance of 1000 m; no specific quantitative measure of the number of tows/hauls taken are provided.

Trawling will likely produce sub-lethal effects (loss of appendages, scarred and broken shells) which will result in an unhealthy residual population, although results are not provided in this report.

6.2.2.2.3 Total direct mortality of invertebrates

Estimates of total direct mortality of invertebrates for organisms caught plus those damaged in the trawl path were calculated based on differences between densities of benthic fauna, pre- and post-trawling. Commercial trawl gear (North Sea: sole trawl gear; Irish Sea: *Nephrops* trawl gear) and towing practices were utilized. Sidescan sonar provided information on seabed topography and impressions of recent trawling activities. T-tests were used to detect statistical differences between mean initial and remaining invertebrate densities. The authors note that reliable estimates of total direct mortality could not be derived for highly mobile migratory epibenthic species such as crabs and shrimp. Total direct mortality was calculated as a function of trawl and sediment.

In the North Sea experiments, invertebrate samples with initial densities of < 5 per 100 m^2 (Triple-D sampling) or less than 10 per m^2 (box core sampling) were not included in the statistical analysis. Mortality estimates derived in the above analysis were then ranked to yield relative vulnerability measures for the invertebrate species; species with the highest mean rank were determined to be the most vulnerable. We note that the possibly most vulnerable species would be those already having densities below 5/100 m^2 and which were not included in the study.

In the Irish Sea, *Nephrops* otter trawl studies were conducted in periodically trawled inshore (35 m) and heavily trawled offshore (75 m) sites. Day grabs were collected to estimate initial and post-trawl densities of invertebrate macrobenthic species. Probability of migration of highly mobile species (e.g., *Nephrops*), and the likelihood of unreliable estimates produced for the proportion of the *Nephrops* population actually in reach of the sampling gear, precluded reliable direct estimates of mortality.

It was noted that silty areas showed less **total directed mortality to burrowing invertebrate** species for otter trawls. This may be due to the fact that otter trawls penetrate less than beam trawls and thus have a lesser effect on the infauna. The most important mortality for invertebrates occurs mainly in the trawl path, possibly as a result of direct physical damage inflicted by the passage of the trawl or indirectly from disturbance and subsequent predation. Fragile or superficial living species showed high mortalities, while robust and deeply burrowing species are less affected. Small individuals were generally less affected.

'Total direct mortality of invertebrates (both discard mortality and mortality in the trawl path as a percentage of initial density) varied for various species of gastropods, starfish, small and medium sized crustaceans, and annelid worms from 10 % to 50 %. For a number of bivalve species, mortalities were found of 30 % to 80 %. Fragile or superficial living species showed high mortalities, robust or deeply burrowing species low or even no mortalities. In general, small sized species and specimens showed relatively low total mortalities.' (IMPACT II, p. 369).

'Otter trawling in silty areas caused less total direct mortality in many burrowing invertebrate species as compared to beam trawling; otter trawls apparently penetrate less deeply into the seabed. Differences in total direct mortalities of benthic fauna due to trawling with 4 m and 12 m beam trawls with tickler chains was slightly higher than for 4 m beam trawls with chain matrices, probably because chain matrices penetrate less deeply into the sediment.' (IMPACT II, p. 369).

'In silty sediments in the North Sea a trend was found for higher total direct mortalities of invertebrates due to beam trawling than in sandy areas. This points to a deeper penetration of beam trawls into a softer seabed.' (IMPACT II, p. 370).

'The species-poor and biomass poor fauna at the offshore station in the Irish Sea illustrates the possible impact of a long-term, high Nephrops trawling effort leading to a species composition that is adapted to a regular fishing disturbance.' (IMPACT II, p. 370).

Although the report and other studies generally support these results and conclusions, some conclusions are not justified. For example, insufficient information precludes the authors' conclusion that long-term high *Nephrops* trawling in the Irish Sea could have resulted in a biomass-poor fauna in the area. Further, if the estimates of total mortality produced by this study can be accepted as valid, they indicate that for most invertebrate species the dominant source of mortality is due to damage caused by the passage of the gear to individuals not retained in the net. These results suggest that the by-catch mortality is a relatively minor proportion of the total mortality, although the quantity of by-catch that is discarded is large relative to the retained catch (10 times). This is a point that is not made strongly by the authors but is consistent with the Section 3.5.1 of the IMPACT II report.

This section of the IMPACT II report is used to support our conclusion that **fragile species are more affected by bottom trawling than robust species** (see Section 6.3, Potential Effects of Bottom Trawls, and below). WGECO felt that the authors did not independently define the fragility or robustness of a species but rather concluded that the species had these characteristics on the basis of the results of the experiment (IMPACT II, Table 3.5.6). Given this circularity of reasoning, there is no conclusive evidence to support this statement provided in this section of the IMPACT II report.

6.2.2.3 Scavenger responses to trawling

Trawling generates dead material, both discards and dead specimens in the trawl track, and Section 3.6 of IMPACT II deals with the response of scavengers to produced offal. The importance of fishery-derived carrion to benthic carnivores was investigated in the North Sea and Irish Sea using a variety of techniques. In the field investigations, the following methods were used: repeated trawling, baited traps, *in situ* observations, and stomach content analysis. The repeated trawling experiments were executed to test the hypothesis that predatory and scavenging fish and invertebrates migrate into recently trawled areas. Before and after experimental trawling, abundance of carnivores was measured, and their stomach contents were analysed. In most studies, densities were also measured in untrawled control strips. Baited (and unbaited) traps were used to identify benthic scavengers that are attracted to specific fishery-related dead materials. For some sites, the area of attraction around the traps was calculated. Scavengers feeding on the dead materials were observed *in situ* by still cameras, video and divers. To examine the rate of discards consumption, fish of known weight were kept on the seabed. The natural rate of decay of dead fish was measured at different seawater temperatures in the laboratory. Stomach contents analyses were done in fish and invertebrates caught at trawled and control areas to reveal the diets of scavenger species. Stomach fullness and diet were established. In the laboratory investigations, feeding and growth of scavenging species were determined, including estimates of length-weight relationships, daily food consumption and measurements of growth. Feeding behaviour of scavenging fish and invertebrates was observed in the laboratory.

'Benthic scavengers and predators feed both on fisheries discards and on animals damaged in the trawl tracks. The responses of scavengers to carrion varies between different sites depending on environmental and physical factors.' (IMPACT II, p. 370).

'In some trawled areas there is opportunistic feeding by a number of predatory species on scavenging species attracted by the disturbance effect. Scavengers can increase their food intake when migrating into and foraging in trawled areas and also alter dietary composition in response to trawling.' (IMPACT II, p. 370).

'Competition for fisheries discards between benthic scavengers sometimes becomes intense and can affect feeding success.' (IMPACT II, p. 370).

'***In the North Sea the annual amount of carrion produced by fishing activities accounts for a maximum of 10 % of annual food consumption by scavenger populations.'*** (IMPACT II, p. 370) [This value was later revised downwards to 7 % (Fonds and Groenewold, in press).]

In addition to the above conclusions, the authors state in the report that migration into trawled areas was observed. Further, they state that the annual amount of carrion generated by the trawling activities does not contribute to an increase in the populations of scavenging fish and invertebrates.

The response of scavengers to fishing disturbance reflects food availability both before and after trawling. The experiment in the Irish Sea was not statistically analysed and so it is not possible to say whether any animal abundance changed, let alone whether they increased or decreased. In the North Sea experiments, no statistical tests were done to determine whether the observed changes were caused simply by the random chances of the sampling process. Nor was there any published indication that the experiments incorporated designs that would allow the effects of trawling to be separated from the natural movements of various species. Indeed, in one case, the sampling gear found extensive material discarded from nearby commercial vessels, which will have provided more food for scavengers than the disturbance caused by the experimental trawls. In the western Irish Sea, appropriate control data were not collected in order to judge whether observed changes were linked to the trawling.

These concerns about the validity of the field experiments are carried through when these numbers are then used to calculate the portion of annual food consumption of scavengers supplied by fishing activities. The validity of extrapolating the data to an annual consumption value, when seasonal variation was not determined, is highly dubious. Even though the final figure is now in the literature (revised downwards to 7 %; Fonds and Groenewold, in press), it should not be extrapolated elsewhere. Despite concerns around this extrapolation, it is recognized that scavengers are likely to react positively to bottom trawl activity.

6.2.2.4 General discussion: fishing mortality in invertebrate populations due to different types of trawl fisheries in the Dutch sector of the North Sea in 1994

This section estimates the annual mortality in 1994 of invertebrate populations caused by different gear types in the southern North Sea (4-m beam trawls, 12-m beam trawls, and otter trawls). The calculations are based on the following information: 1) estimates of total mortality due to a single trawl pass reported in earlier sections, 2) survey densities of benthic invertebrates by ICES rectangle (Bergman and van Santbrink, 1997), 3) fishing effort by ICES rectangle, 4) the concentration profile of beam trawl effort within an ICES rectangle based on effort micro-distribution data from a subset (n = 25) of the Dutch fleet (Rijnsdorp *et al.*, 1997). The calculations do not consider intraseasonal recruitment or sources of mortality other than trawling (i.e., natural mortality).

'In the Dutch sector in 1994, the 12 m beam trawl fishery was the dominant type of trawling offshore, with an average frequency of 1.23. The average frequency of the 4 m beam trawl fishery with ticklers 0.13 mainly in the coastal zone, that of 4 m beam trawl fishery with chain matrices was 0.01 exclusively in the southernmost areas, and that of the otter trawl fishery was 0.06.' (IMPACT II, p. 371).

'The annual fishing mortality in the larger sized invertebrate populations varied from 7 % to 48 % due to trawl fisheries in the Dutch sector in 1994, with half the number of species showing values of >25 %. The 12 m beam trawl fisheries caused higher fishing mortalities than 4 m beam trawl and otter trawl fisheries. Only in species restricted to the coastal zone, where 4 m beam trawl fishery is much more intensive than in offshore areas, fishing mortalities due to this fishery were relatively higher and might even exceed that due to 12 m beam trawl fishery.' (IMPACT II, p. 371).

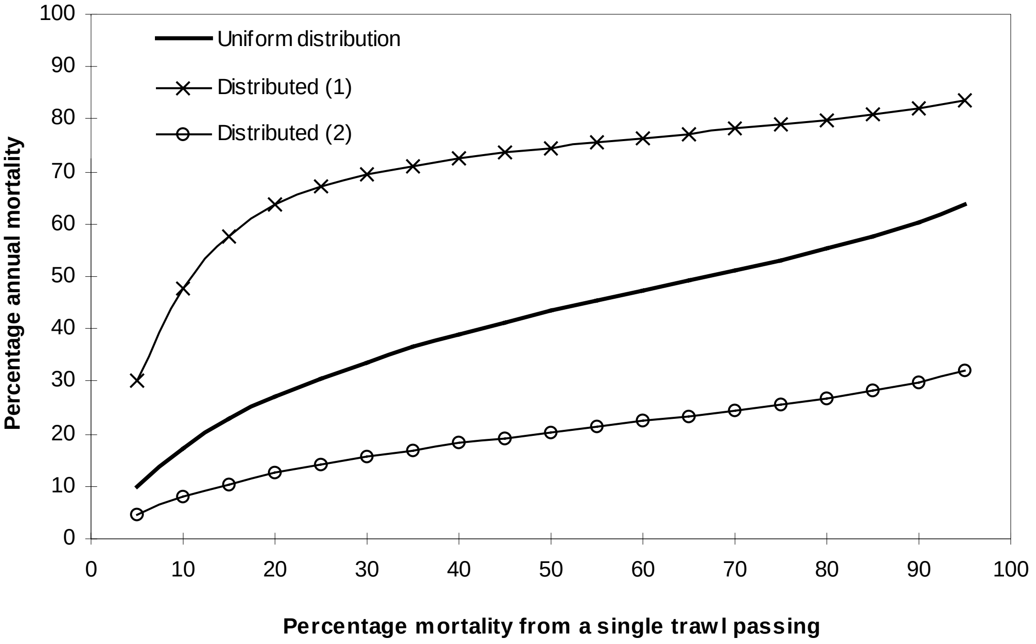
These estimates of annual mortality should be considered rough approximations. The assumptions required to obtain such estimates without suitably detailed information on the spatial distribution of benthic invertebrates and on the spatial distribution of fishing effort were reasonable and appropriate. However, estimates depend heavily on those assumptions, which cannot be evaluated independently. For example, the concentration profile of beam trawl effort within an ICES rectangle was data-based, but a uniform spatial distribution was assumed for invertebrate species. An alternative assumption would be of an invertebrate species having a low and uniform distribution across an ICES rectangle except for a high abundance (60 % of total abundance in the area) in the 1/9th which is most heavily trawled (56.9 % of the effort). This assumption is hypothetical because any slow-moving invertebrate species, with any substantial mortality due to trawling, would have already been extirpated or reduced to very low abundance in the most heavily fished areas (Distribution 1 on Figure 6.2.2.4.1). The opposite example (Distribution 2 on Figure 6.2.2.4.1), which assumes that 60 % of the total abundance is located in the 1/9th of the area which is least fished (0.1 % of the effort), is also displayed in Figure 6.2.2.4.1. Both cases are compared in Figure 6.2.2.4.1 with the mortalities derived using a uniform distribution of the species as done in IMPACT II. This figure shows that estimates of annual mortality are highly dependent on which distribution is assumed to be most appropriate for a particular species (Figure 6.2.2.4.1).

In addition, estimates of annual mortality depend on total mortality estimates reported in Section 3.5 of IMPACT II (which are highly uncertain). The annual fishing mortality estimates of benthic invertebrates ranged from 7 % to 33 % and are highly affected by the assumption of the type of distributions of the species and the fisheries within the rectangles.

6.2.2.5 Evidence in support of the effects of bottom trawls

This section relates the conclusions found in Sections 3.4 to 3.6 of IMPACT II and discussed above (collectively referred to as Mortality by WGECO) to Section 6.3, below, that summarises the effects of bottom trawls on species. References to the wider literature on these effects are provided where appropriate. The Mortality section of IMPACT II, and the other literature on that topic, were not expected to contain information on the effects of bottom trawling on habitats.

Figure 6.2.2.4.1. A comparison of the effect on annual mortality using different spatial distributions of an invertebrate species relative to fishing effort. The overall annual effort is set to 2.4 trawl passings (similar to a heavily trawled rectangle). The calculations do not consider recruitment nor sources of mortality other than bottom trawling.



6.2.2.5.1 Species composition

Bottom trawling can cause a reduction in the geographic range of species

Sections 3.4 to 3.6 of IMPACT II do not address this issue. Reference to other literature that supports this conclusion is provided in Section 6.2.4.5, below.

Bottom trawling can cause a decrease in populations that have low rates of turn-over

Sections 3.4 to 3.6 of IMPACT II do not address this issue because it was conducted in an area where this type of damage had already occurred. Reference to other literature is provided in Section 6.2.4.5, below.

Bottom trawling is patchy and can cause fragmentation of populations

Sections 3.4 to 3.6 of IMPACT II do not address this issue. Reference to other literature is provided in Section 6.2.3.7 of this report.

Fragile species are more affected by bottom trawling than robust species

We can find no conclusive evidence to support this statement provided in these sections of the IMPACT II report. The vulnerability of bivalve molluscs to damage can be ranked according to the mechanical shell strength and it has been demonstrated that thin-shelled bivalves are more easily damaged by otter trawl doors than thick-shelled bivalves (Rumohr and Krost, 1991). However, Gilkinson *et al.*, (1998) have also shown that certain size classes within species are protected from the gear through displacement in the fluidized sediment ahead of the trawl doors. Therefore, this statement is too general to be classified as a general effect without qualification. Description of the classifications and qualifiers as to size would be necessary. However for upright brittle species, such as hard corals, this statement would apply. Further reference to other literature is provided in Section 6.2.3.7 of this report.

Surface-living species are more affected by bottom trawling than deep-burrowing species

This is supported in this section and is a direct consequence of the penetration of the gear. Reference to other literature is provided in Section 6.2.3.7 of this report.

The relative abundance of species is altered by bottom trawling

There is limited indirect evidence (mechanisms which could lead to this result) to support this conclusion in the section of the IMPACT II report dealing with catch efficiency for small-sized fish and invertebrates (IMPACT II, Section 3.4), however no analyses were performed with the data and direct evidence is considered elsewhere in this report (Section 6.2.3.6).

Bottom trawling can have sub-lethal effects on individuals

There is only limited evidence for this in the IMPACT II report. Various degrees of damage on epibenthic and shallow infaunal bivalves have been reported for different types of mobile fishing gear (e.g., Kaiser and Spencer, 1996).

Bottom trawling can cause an increase in populations that have high rates of turn-over

Sections 3.4 to 3.6 of IMPACT II do not address this issue. Some reference to other literature is provided in Section 6.2.3.7 of this report.

Bottom trawling favours populations of scavenging species

The role of scavengers in marine ecosystems perturbed by human activities has been reviewed by Britton and Morton (1994). Although the IMPACT II report does not provide conclusive evidence of this phenomenon, other studies support this conclusion. For example, Kaiser and Spencer (1994) have shown that fish aggregate in the track of a beam trawl (35 times more fish were found in the track of the beam trawl than in adjacent reference areas). This is expected to be a very short-term effect for fish. Whiting were found to have eaten the gonads of Queen scallops, a prey type that is not normal for them, suggesting that the bivalves had been damaged by the trawl. Invertebrate scavengers also aggregate (for a somewhat longer period) and alter their feeding behaviour in areas disturbed by trawling, although the response is variable (e.g., Ramsay *et al.*, 1998; Prena *et al.*, 1999) and likely to depend upon the points noted above in Section 6.2.2.3. Evidence for longer-term impacts of trawling on scavengers is detailed in Section 6.2.4.2.2. Additional comments are found in Section 6.2.3.6 of this report.

6.2.2.5.2 References

Anon. 1999. Survival of fish and Nephrops. Final Report of European Commission Contract FAR-CT95-0753.

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

Britton, J.C., and Morton, B. 1994. Marine carrion and scavengers. Oceanography and Marine Biology: an Annual Review, 32: 369–434.

Erickson, D., Pikitch, E., Suuronen, P., Lehtonen, E., Bublitz, C., Klinkert, C., and Mitchell, C. 1999. Selectivity and mortality of walleye pollock escaping from the codend and intermediate (=extension) section of a pelagic trawl. Final report submitted to NMFS by Alaska Fisheries Development Foundation, Anchorage, Alaska.

Fonds, M., and Groenewold, S. 1999. Food generated by beam trawl fishery for sole in the southern North Sea, for benthic carnivores and demersal fish. *In* Effects of fishing on non-target species and habitats. Biological conservation and socio-economic issues. Ed. by M. Kaiser and S.J. de Groot. Blackwell Science, Oxford. 416 pp.

Fréchet, A. 1996. Intercalibration de huit chalutiers participant aux pêches sentinelles dans le nord du golfe du Saint-Laurent en 1995 à l'aide de sondes SCANMAR. Intercalibration of eight otter-trawlers participating in the sentinel fisheries in the Northern Gulf of St. Lawrence (3Pn,4RS) in 1995 through the use of SCANMAR sensors. DFO Atlantic Fisheries Research Document 96/67. 15 pp.

Fréchet, A. 1997. Standardisation de chalutiers participant aux pêches sentinelles dans le nord du golfe du Saint-Laurent en 1996. Standardization of otter trawlers participating in the sentinel fisheries in the Northern Gulf of St. Lawrence in 1996. DFO Atlantic Fisheries Research Document 97/72. 10 pp.

Gilkinson, K., Paulin, M., Hurley, S., and Schwinghamer, P. 1998. Impacts of trawl door scouring on infaunal bivalves: results of a physical trawl door model/dense sand interaction. Journal of Experimental Marine Biology and Ecology, 224: 291–312.

Kaiser, M.J., and Spencer, B.E. 1994. Fish scavenging behaviour in recently trawled areas. Marine Ecology Progress Series, 112: 41–49.

Prena, J., Schwinghamer, P., Rowell, T.W., Gordon, D.C., Gilkinson, K.D., Vass, W.P., and McKeown, D.L. 1999. Experimental otter trawling on a sandy bottom ecosystem of the Grand Banks of Newfoundland: Analysis of trawl bycatch and effects on epifauna. Marine Ecology Progress Series, 181: 107–124.

Ramsay, K., Kaiser, M.J., and Hughes, R.N. 1998. Responses of benthic scavengers to fishing disturbance by towed gears in different habitats. Journal of Experimental Marine Biology and Ecology, 224: 73–89.

Rijnsdorp, A.D., Buijis, A.M., Storbeck, F., and Visser, E. 1997. De microverspreiding van de Nederlandse boomkorvisserij gedurende de periode van 1 april 1993 tot en met 31 maart 1996. RIVO-DLO rapport:C0006/97. 39 pp.

Rumohr, H., and Krost, P. 1991. Experimental evidence of damage to benthos by bottom trawling with special reference to *Arctica islandica*. Meeresforschung, 33: 340–345.

6.2.3 Comparison of disturbed and undisturbed areas

6.2.3.1 Introduction

Benthic communities were studied at three locations where trawled and untrawled areas were compared. These included two areas where trawling was prevented by seabed wreckage (discussed in Sections 6.2.3.3. and 6.2.3.4, below) and an area that had been closed to fishing (discussed in Section 6.2.3.2, below). In the latter, the fishing gear used comprised only a set of otter-boards wires and ground gear but no net. In this respect it did not represent fishing operations. Studies focusing on comparing disturbed versus undisturbed areas have their specific difficulties (see Underwood, 1997), which include finding undisturbed areas, locating matched areas which differ only for the factor under consideration and analytical problems such as pseudoreplication (Hurlbert, 1984).

These three studies are discussed separately on the basis of study design and (statistical) interpretation of the results. Study-specific conclusions are addressed in each subsection. The general conclusions drawn from the IMPACT II study report (p. 370) will be discussed in Section 6.2.3.5, below. In Section 6.2.3.6, these are linked to the generic list of potential effects discussed in Section 6.3.

6.2.3.2 Loch Gareloch

A field experiment to test the impacts of trawling was conducted in Loch Gareloch (Scotland), in an area that had been closed to fishing for over 25 years. However, no net was used in the fishing experiment, so the experiment does not duplicate all the effects of fishing. Benthic communities were compared in a treatment plot and a reference unfished plot. Treatment plots were trawled once a month, on each occasion 10 times, over period of 16 months, after which trawling ceased. Samples were collected at regular intervals prior to trawling, during the 16 months of trawling, and for 18 months after trawling had ceased. According to the findings, the abundances of several species, the median particle diameter, and organic carbon content differed significantly between treatment and control plots initially. Abundances of infauna following trawling increased (mainly polychaetes), decreased, or did not show any obvious changes. On the multidimensional scaling (MDS) ordination, communities in treatment and control plots became more distinct following trawling, with some indication that they became more similar after the trawling activity ceased. Some effects on epifauna were found though epifaunal densities were low and impacts on epifauna appeared to be short lived.

Comments on results

'At the sheltered sealoch study area, trawling disturbance had a clear effect, increasing the numbers of species and numbers of individuals, and decreasing diversity. Certain opportunistic species (mainly small polychaetes), considered to be indicators of disturbance, became more abundant in the treatment area when compared to the reference area, both during and following the disturbance period.' (IMPACT II, p. 260).

No species were found to have significantly increased in abundance in a consistent pattern following trawling. Several species were however found in greater abundances within treatment plots prior to trawling and remained so throughout the experiment. The suggested increase in abundance of several species within treatment plots was attributed to the increase in the numbers of opportunist species. Although opportunist species might be expected to show rapid increase in numbers (Grassle and Saunders, 1973) following disturbance, followed by a decline, no species showed such patterns, except perhaps the polychaetes *Caullierella zetlandica* and *Pseudopolydora paucibranchiata*.

'The densities of some species declined relative to the reference area, suggesting these species may be sensitive to physical disturbance.' (IMPACT II, p. 260).

No species showed a consistent decline in numbers following trawling within treatment plots relative to control plots. For example, the species *Nucula nitidosa* and *Nepthys cirrosa*, which were found in significantly lower abundances within treatment plots compared to control plots following trawling on month 5 and 10, did not differ after 16 months of trawling activity.

Considering the experimental design and the statistical analysis employed, many of the conclusions may not be convincing due to the fact that prior to trawling the benthic communities in the treatment and control plots were different. Median particle diameter and percentage of organic material differed between plots prior to trawling and remained. Since the structure of benthic communities depends to a large extent on sediment characteristics, this may have accounted for the observed patterns. The experiment is pseudoreplicated as only two plots were compared, but due to the small size of the loch, multiple control and treatment plots could not be established, and this is discussed in a subsequent publication (Tuck *et al.*, 1998). Using two-way ANOVA for the analysis of these data is inappropriate as it is only possible to compare differences between the two locations, but not to attribute them to trawling effects. The authors argue that differences in faunal abundances through time between treatment and control plots (i.e., significant interaction terms) were due to the effects of trawling. These changes were not consistent among species and differences in temporal patterns between plots were not convincing for several species. Results from other studies of otter-trawling impacts on the smaller infauna that used replicated designs (Brylinski *et al.*, 1994; Lindegarth, in press; see also Gilkinsson *et al.*, 1998) have so far found very few effects, although some effects have been found on epifauna (Collie *et al.*, 1997; Prena *et al.*, 1999). Effects of beam trawling on the smaller infauna appear to be greater and beam trawls have been found to alter abundances of small infauna significantly (e.g., Kaiser and Spencer, 1996).

6.2.3.3 West Gamma wreck study

West Gamma is a wreck of an oil drilling platform, which sank in 1990 in the outer German Bight northwest of Helgoland. Fishing was prohibited within an area marked by buoys in the vicinity of the wreckage. In 1994, part of the wreck was removed as well as the buoys. Prior to the removal of the buoys, benthic samples were collected inside and outside the protected area. Sampling continued for one year after the buoys were removed. Two sampling devices were used, a van Veen grab for the smaller infauna and a small dredge for the larger and mobile fauna.

Over the study period, there were considerable seasonal fluctuations in species abundance. Total numbers of species and individuals increased between 1992 and 1994 and remained similar from 1994 onwards.

'The results of the West Gamma study show clear differences in the macrozoobenthic community of the area enclosed by the buoys and the surrounding unprotected area' (IMPACT II, p. 260).

'The reduction of detectable effects to the immediate vicinity of the wreck following removal of the buoys underlines that the differences detected earlier were related to the protection of the area by the buoys.' (IMPACT II, p. 260).

Faunal abundances, composition, and diversity within and outside the protected area did vary greatly between years but differences were not consistent. In August 1994, diversity (Shannon's) and evenness (Pileou) were greater in the fished area; they were similar in both areas in April 1994, but in September 1992, the diversity was lower in the fished area. The total number of species at each sampling occasion increased from 1992 and onwards, being similar in the fished and the unfished area with the exception of samples from April 1994. These results, therefore, conflict with the trends in diversity described above. The sediment analyses suggest a difference between the areas. There can be at least two treatment effects: the presence of the wreck influencing the environment, while trawling affects the composition of the benthos. Following the removal of the buoys, species composition at some stations which were previously located within the protected area may have become somewhat closer in similarity to those from the fished area, possibly because the fishing activity had moved closer to the wreck. Removal of part of the wreck in 1994 would, without doubt, result in transport and suspension of sediments and so may have had further impact on fauna. No information on fishing effort in the area is reported. A similar study (Hall *et al*., 1993) showed that differences in faunal abundances in areas close to wrecks and adjacent areas were well correlated with median particle diameter of the sediments, but could not be attributed to patterns in the fishing effort.

6.2.3.4 Iron Man and 41 Fathom Fast

The distribution of benthic fauna in the vicinity of two shipwrecks was analysed. The two wrecks (known as Iron Man and 41 Fathom Fast) were located on *Nephrops* fishing grounds in the Irish Sea. The 41 Fathom Fast area was considered to be more heavily trawled than the Iron Man area (p. 45). The water depth at the Iron Man wreck and the 41 Fathom Fast wreck was about 35 m and 75 m, respectively, and sediments consisted of muddy-sand and sandy silt, respectively. The areas around the wrecks were sampled with van Veen grabs on three transects radiating out from the wrecks. These transects extended out to a distance of between 125 m to 400 m from Iron Man, and from 125 m to 500 m from 41 Fathom Fast. At each transect, three stations were sampled. Sampling was carried out in May 1995 in the Iron Man area and in April 1996 in the 41 Fathom Fast area. Fauna in samples from transects were compared with samples from nearby fishing grounds before and after experimental trawling.

Comments on results

'At the lower intensity fishing area, the species rich fauna of the 'unfished' wreck site may resemble the natural undisturbed fauna characteristic of the region, prior to commencement of the Nephrops fishery. In comparison to this protected area the fauna at the inshore station showed some indication of disturbance (reduction in number of species and biomass, disappearance of some fragile bodied species), but still included some large molluscs and had similar diversity to the wreck area. Multivariate analysis showed clear differences between the fished and unfished areas.' (IMPACT II, p. 261).

In the Iron Man area, the total number of individuals and biomass (g m^{-2}) differed between samples taken from the transect (adjacent to the wreck) and from the fishing grounds. Sample variances are not given and no attempt has been made to compare differences statistically. No information on spatial distribution of the fishing effort and trawling intensity is given, and therefore it is doubtful to conclude that differences in faunal composition and abundances in the two areas are due to trawling impacts. Differences in the number of species, biomass, species richness and Shannon's diversity index before and after experimental trawling at the fishing ground may indicate (although not tested statistically) that trawling may have had short-term impacts.

A prerequisite for interpreting studies on the impact of bottom trawling on benthic ecosystems is the location of control areas which are not affected by trawling. Resuspensions of bottom materials, such as sediments and organisms, during the trawling process may be transported and settle outside the trawled area (Currie and Parry, 1996), which may bias experiments where the control sites are just outside the treated site (Brylinsky *et al.*, 1994). In other words, it may well be that the unfished areas around the wreck are affected by the trawling activities in the adjacent areas.

'At the heavily fished site the biomass and diversity and average size of individuals all appeared greater in the protected area than the surrounding ground [fished area]' (IMPACT II, p. 261).

In the 41 Fathom Fast wreck area, there were some indications of differences between transects and the fishing ground. All community parameters were lower at the fished ground both before and after experimental trawling, but this was not tested statistically. Furthermore, MDS plots indicated that samples from the transects and the fishing grounds formed separate groups. The validity of comparing the results from the transects and the fishing grounds is unclear, because due to lack of information about the spatial distribution of the fishing effort in the fishing grounds it is difficult to interpret these results.

'While both sites showed considerable disturbance effect, this effect appeared to be greater at the higher intensity fishing area, and impact may therefore be related to intensity of fishing.' (IMPACT II, p. 261).

Locations not previously exposed to demersal fishing activities have demonstrated larger short-term ecosystem effects than frequently fished grounds (Pranovi and Giovanardi, 1994), whereas no experimental study has focused on the relation between fishery intensity and long-term changes of the ecosystem. In the IMPACT II report, the greater long-term impact is observed in the heavily fished area and a lower impact is observed in the less heavily fished area. They conclude that the impact is related to fishing intensity. These two study sites were very different according to supposed fishing intensity, but also differed in water depth, sediment composition, and biotic community parameters. Sediment texture is reported to be a covariate with the effect of trawling and dredging (Shephard, 1983; Kaiser and Spencer, 1996). Furthermore, the sampling of the areas was conducted in two different years. The comparison of the community composition at the two study areas may then be biased by both spatial and temporal variability. The conclusion that impact may be related to fishing intensity is reasonable, but this study provides little evidence for that proposition.

6.2.3.5 General conclusions from the IMPACT II Report on Chapter 3.7 (p. 370)

The results of the three individual studies were used to draw up some general conclusions. It was not always clear which results were considered by the authors to support a particular conclusion. Therefore, we could not assign each conclusion to a specific study and we consider this set of conclusions in this separate section.

'Experimental disturbance of a previously unfished site showed clear long term effects on both epi- and infauna. Comparison of fished and protected sites within fishing grounds also showed clear differences, suggesting that fishing disturbance has significant long term effects on benthic communities' (IMPACT II, p. 370).

Specific difficulties include the pseudo-replication and significant pre-impact differences at the Gareloch site, the lack of statistical power in the Irish Sea studies, and the lack of data on fishing effort at the West Gamma study. The results presented in IMPACT II do provide some indication of demonstrable changes in the benthos and these, taken alongside the growing evidence from the literature, provide a convincing body of evidence for trawling impacts on benthic communities (e.g., Kaiser and Spencer, 1996; Prena *et al.*, 1999).

'Comparison of the two Irish Sea sites showed an increasing effect of fishing with greater fishing intensity. At the heavily fished site the fauna already acclimated to intense fishing disturbance and no short-term effects could be detected with the sampling methods used ' (IMPACT II, p. 370).

The nature of the change in fishing effort and the magnitude of the change along the presumed gradients of fishing intensity are not quantified. Therefore, it is impossible to conclude that a monotonic relationship exists between fishing intensity and benthic effects. Essentially all conclusions follow from basic ecological principles and logic, not from detailed quantitative investigations. When an unimpacted area is first trawled, there is almost certainly an effect and the relationship between magnitude of effect and intensity of fishing is likely to be monotonic. For a specific site, it is highly unlikely that increased trawling intensity ever reduces impact relative to some lesser intensity of trawling. At high levels of disturbance, an asymptote is expected such that beyond a threshold intensity of trawling, further trawling has negligible incremental effects. Beyond being monotonic, and likely to have an asymptote somewhere on the impact axis, very little can be generalized about the functional relationship of impact and effort. The shape will be very dependent on specific characteristics of the area being fished—the initial biological diversity, structural complexity, history of disturbance by other (natural and anthropogenic) factors, etc. The shape of the relationship will also depend on the spatial scale being considered. The above argument is developed specifically for the path of the trawl. As soon as one considers areas larger than a trawl path being repeatedly fished, the distribution of trawl events within the area has a great effect on the shape of the relationship between effort and impact.

'In general, opportunistic (small size, fast reproducing) species increased in abundance while sensitive (large size, fragile) species declined in numbers due to trawling disturbance. The ability of mobile scavengers to migrate in and out of disturbed areas makes the detection of trawling effects on these species difficult.' (IMPACT II, p. 370).

It is critically important that the examination of patterns of response by life history traits are based on the allocation of taxa to life history groups by criteria which are derived independently of the data. These criteria should form a basis for hypothesis generation. We are aware of the depressingly few studies which have addressed fishing impacts on benthic organisms using *a priori* classification of the fauna into life history groups and the responses of these life history groups prior to the application of the treatment (e.g., Frid and Hall, 1999). The Loch Gareloch study suffers from the common problem of observing a response and then providing *a posteriori* explanations. No direct evidence is presented of species increasing in abundance in response to bottom trawling. Some species were more abundant outside the wreck areas, but these may be site effects.

The study designs of the Loch Gareloch and West Gamma studies do not allow any comments to be made on the abundance of scavengers.

Measures of diversity and evenness were consistently higher in unfished areas when compared to adjacent disturbed areas.' (IMPACT II; p. 370).

In Loch Gareloch species richness, the most direct measure of species diversity, was higher in the fished area. The pattern for other diversity measures is more complex but the application of parametric tests (e.g., ANOVA) to these measures are not appropriate because the underlying structure of such indices rarely meet the assumptions of the tests. In the Irish Sea, diversity (S: species richness, H′: Shannon's) was lower in the fished areas in only 5 of the 8 pairwise comparisons. Therefore, these results do not support the conclusion above.

It is generally accepted that ecological communities show a dome-shaped response in diversity to a wide variety of perturbations, with maximum diversity occurring at some intermediate levels of disturbance. The exact parameters defining the dome are highly community specific. Without prior knowledge of the disturbance régime operating on a particular community, it is not possible to predict the trajectory of the response in the face of additional impacts.

'Results from the Loch Gareloch study suggest that in sheltered muddy sites, recovery following disturbance may take over 18 months. In regularly fished area, communities may never fully recover before being redisturbed.' (IMPACT II, p. 370).

Estimates of recovery times following anthropogenic disturbances are important for managing marine ecosystems. Given the lack of demonstrable treatment effects, it is not possible to make any estimate of recovery time from these data. It is widely accepted that the majority, if not all, marine communities are open, non-equilibrium systems. As such, they will continually change in response to a host of intrinsic and extrinsic factors, including natural and anthropogenic perturbations. This report provides no additional evidence for this, but it remains true that marine ecosystems are open non-equilibrium systems. We are therefore concerned about the perception that if fishing stops an area will recover to something akin to the unimpacted state. Given the open nature of marine communities, this could only happen if a suitable source of propagules existed nearby, possibly in roughly the balance of the pre-impacted community, and if hydrographic and climatic conditions had not changed.

We are aware of the literature containing many case studies which show that the rate of recovery of benthic communities is subject to many case-specific factors and for impacts occurring on large spatial scales, recovery estimates of several years are not uncommon (see Hall and Raffaelli, 1994). Furthermore, it is well known that disturbed communities often do not recover to the original state. Communities are in a continual state of flux and unimpacted areas change over time, while recovering sites may follow a successional trajectory which leads to a different community state (Hall and Raffaelli, 1994; Thrush *et al.*, 1996).

6.2.3.6 Generic potential effects supported by examples of 'Disturbed versus undisturbed areas'

6.2.3.6.1 Habitats

Bottom trawling can remove some physical features

Physical features were not considered in these portions of the IMPACT II report (see Sections 6.2.3.2, 6.2.3.3 and 6.2.3.4, above). Other studies have demonstrated that physical features such as cobbles and boulders are moved by trawling (Freese *et al.*, 1999).

Bottom trawling can cause a reduction in structural biota (biogenic features)

Studies were principally carried out in soft sedimentary environments where such biogenic features are not found (Sections 6.2.3.2, 6.2.3.3, 6.2.3.4, above, Chapter 3.7 in the IMPACT II report). Effects of trawling have been demonstrated in habitats characterized by biogenic features such as sponges (Poiner *et al.*, 1998), eelgrass (van Dolah *et al.*, 1987), worm tubes, bryozoa and hydrozoa (Bradstock and Gordon, 1983; Collie *et al.*, 1997) and corals (Fosså, in press).

Bottom trawling can cause a reduction in complexity

The results reported in IMPACT II (see Sections 6.2.3.2, 6.2.3.3, and 6.2.3.4, above) provide no evidence for such effects as the habitats in these studies cannot be regarded as complex, possibly due to long history of fishing. Trawling is known to reduce the complexity of heterogeneous habitats, especially if they are dominated by fragile epifauna (Freese *et al.*, 1999).

Bottom trawling can alter the detailed physical structure of the sea floor

Sediment characteristics in trawled plots within Loch Gareloch (see Section 6.2.3.2, above) may have changed as a result of trawling. The problems with the experimental design (as discussed in Section 6.2.3.2, above) make it difficult to definitely attribute changes in sediment characteristics to trawling impacts. The sediments within the West Gamma wreck area (see Section 6.2.3.3, above) differed from the surrounding sediments. There was no strong evidence in Chapter 3.7 of the IMPACT II report to support the assertion that these differences were due to trawling. Kaiser and Spencer (1996) demonstrated that beam trawling can change the physical characteristics of the surface sediments.

6.2.3.6.2 Species

Bottom trawling can cause the loss of species from part of their normal range

The studies in the IMPACT II report (see Sections 6.2.3.2, 6.2.3.3, and 6.2.3.4, above) provide no evidence for this.

Bottom trawling can cause a decrease in populations which have low rates of turnover

In all three studies reported in the IMPACT II report (see Sections 6.2.3.2, 6.2.3.3, and 6.2.3.4, above), the majority of species had a high population turnover. For those species with a low population turnover, the data provided no evidence that they had been affected due to trawling activities. Bergman and Hup (1992) found a decrease in species with low turnover.

Bottom trawling can cause fragmentation of populations

Studies reported in Chapter 3.7 of the IMPACT II report (see Sections 6.2.3.2, 6.2.3.3, and 6.2.3.4, above) provided no evidence for fragmentation of populations due to the small scale of undisturbed areas. Trawling has impact on deep-water corals (*Lophelia*) around the coast of Norway by fragmenting the spatial distributions of these corals (Fosså, in press).

The relative abundance of species is altered by bottom trawling

In the Loch Gareloch study, differences were found between treatment and control plots (Chapter 3.7 of the IMPACT II report; see Section 6.2.3.2, above), but as the experiment was pseudo-replicated, these changes cannot be unambiguously attributed to trawling. Other studies have found changes in abundances (generally a decrease) of infauna (Kaiser and Spencer, 1996).

Fragile species are more affected by bottom trawling than robust species

In the Loch Gareloch study, it was stated that several fragile species such as *Metridium senile* declined in abundance due to trawling (see Section 6.2.3.2, above). It is clear that in Loch Gareloch, there are few species which can be regarded as fragile, as acknowledged by the authors (Tuck *et al*., 1998), such as corals and sponges which are known to be especially susceptible to disturbance (e.g., Freese *et al*., 1999). It is not clear whether *Metridium senile* is as vulnerable to trawling disturbance as claimed by the authors. In the wreck studies (see Sections 6.2.3.3. and 6.2.3.4, above), none of the species recorded there can be regarded as fragile.

Surface living species are more affected by bottom trawling than deep-burrowing species

In the Loch Gareloch study (see Section 6.2.3.2, above) and the wreck studies (see Sections 6.2.3.3. and 6.2.3.4, above), the majority of the fauna are small and probably easily suspended by the trawl and redistributed without harm. Of the large surface-living species, some could be affected by bottom trawls, however the studies reported in Chapter 3.7 of the IMPACT II report provide little evidence as to whether that is the case. Bergman and Hup (1992) measured higher mortality rates for surface-dwelling benthos than for deep-burrowing fauna.

Bottom trawling can have sub-lethal effects on individuals

No evidence from the studies reported in Chapter 3.7 of the IMPACT II report suggest that trawling has sub-lethal effects on populations. Kaiser (1996) showed that trawling had sublethal effects on two species of starfish.

Bottom trawling can cause an increase in populations that have high rates of turn-over

In the Loch Gareloch study (Chapter 3.7 of the IMPACT II report), the authors concluded that several species of polychaetes became very abundant in treatment plots following trawling. This was refuted in Section 6.2.3.2, above. We are not aware of studies that have found increases of populations with high rates of turn-over within trawled areas.

Bottom trawling favours populations of scavenging species

The role of scavengers is not discussed in the Loch Gareloch study (Chapter 3.7 of the IMPACT II report and Section 6.2.3.2, above, but see Tuck *et al.,* 1998). In the West Gamma study (see Section 6.2.3.3 above), greater number of scavengers were found in the near vicinity of the wreck compared to adjacent fished areas, but it is not possible to attribute these differences to trawling. Invertebrate and fish scavengers are known to enter areas which have been recently trawled (Kaiser and Spencer, 1994; Kaiser and Ramsay, 1997).

6.2.3.7 References

Bergman, M.J., and Hup, M. 1992. Direct effects of beamtrawling on macrofauna in sandy sediments of the southern North Sea. ICES Marine Science Symposia, 49: 5–11.

Bradstock, M., and Gordon D.P. 1983. Coral-like bryozoan growths in Tasman Bay, and their protection to conserve commercial fish stocks. New Zealand Journal of Marine and Freshwater Research, 17: 159–163.

Brylinsky, M., Gibson, J., and Gordon, D.C. 1994. Impacts of flounder trawls on the intertidal habitat and community of the Minas Basin, Bay of Fundy. Canadian Journal of Fisheries and Aquatic Sciences, 51: 650–661.

Collie, S., Escanero, G.A., and Valentine, P. 1997. Effects of bottom fishing on the benthic megafauna of Georges Bank. Marine Ecology Progress Series, 155: 159–172.

Currie, D.R., and Parry, G.D. 1996. Effects of scallop dredging on a soft sediment community: a large-scale experimental study. Marine Ecology Progress Series, 134: 131–150.

Fosså, J.H. In press. The magnificent coral reefs in Norway and the destructive works of the trawling fleet. ICES Symposium on Marine Benthos Dynamics: Environment and Fisheries Impacts.

Freese, L., Auster, P.J., Heifetz, J., and Wing, B.L. 1999. Effects of trawling on seafloor habitat and associated invertebrate taxa in the Gulf of Alaska. Marine Ecology Progress Series, 132: 119–126.

Frid, C.L.J., and Hall, S.J. 1999. Inferring changes in North Sea benthos from fish stomach analysis. Marine Ecology Progress Series, 184: 183–188.

Gilkinsson, K., Paulin, M., Hurley, S., and Schwinghamer, P. 1998. Impacts of trawl door scouring on infaunal bivalves: results of a physical trawl door model/dense sand interaction. Journal of Exploration Marine Biology Ecology, 224: 291–312.

Grassle, J.F., and Saunders, H.L. 1973. Life histories and the role of disturbance. Deep Sea Research. 20: 643–659.

Hall, S.J., and Raffaelli, D. 1994. Patchiness and disturbance in shallow water assemblages. *In* Aquatic Ecology, Scale, Pattern and Process, pp. 333–375. Ed. by P.S. Giller, A.G. Hildrew, and D.G. Raffaelli. Blackwell Scientific Publications, Oxford.

Hall, S.J., Robertson, M.R., Basford, D.J., and Heaney, S.J. 1993. The possible effects of fishing disturbance in the northern North Sea: an analysis of spatial patterns in community structure around a wreck. Netherlands Journal of Sea. Research, 31: 201–208.

Hurlbert, S.H., 1984. Pseudoreplication and the design of ecological field experiments. Ecological Monographs, 54: 187–211.

Kaiser, M.J. 1996. Starfish damage as an indicator of trawling intensity. Marine Ecology Progress Series, 143: 303–307.

Kaiser, M.J., and Ramsay, K. 1997. Opportunistic feeding by dabs within areas of trawl disturbance: possible implications for increased survival. Marine Ecology Progress Series, 152: 307–310.

Kaiser, M.J., and Spencer, B.E. 1994. Fish scavenging behaviour in recently trawled areas. Marine Ecology Progress Series, 112: 41–49.

Kaiser, M.J., and Spencer B.E., 1996. The effects beam-trawl disturbance on infaunal communities in different habitats. The Journal of Animal Ecology, 65: 348–358.

Lindegarth, M., Hansson, M., Valentinsson, D., and Ulmesterand, M. In press. Disturbance by trawling changes temporal and spatial structure of benthic-communities in Gullmarsfjorden, Sweden. ICES Symposium on Marine Benthos Dynamics: Environmental and Fisheries Impacts.

Poiner, I., Glaister, J., Pitcher, R., Burridge, C., Wassenberg, T., Gribble, N., Hill, B., Blaber, S., Milton, D., Brewer, D., and Ellis, N. 1998. Final report on effects of trawling in the Far Northern section of the Great Barrier Reef: 1991–1996. CSIRO Division of Marine Research, Cleveland.

Pranovi, F., and Giovanardi, O. 1994. The impact of hydraulic dredging for short-necked clams, *Tapes spp.*, on an infaunal community in the lagoon of Venice. Scientia Marina, 58: 345–353.

Prena, J., Schwinghamer, P. Rowell, T.W. Gordon, D.C., Gilkinsson, K.D., Vass, W.P., and McKeown, D. 1999. Experimental otter trawling on a sandy bottom ecosystem of the Grand Banks of Newfoundland: analysis of trawl bycatch and effects on epifauna. Marine Ecology Progress Series, 181:107–124.

Shephard, S.A. 1983. The epifauna of megaripples: species' adaptations and population responses to disturbance. Australian Journal of Ecology, 8: 3–8.

Thrush, S.F., Whithlatch, R.B., Pridmore, R.D., Hewitt, J.E., Cummings, V.J., and Wilkinson, M.R. 1996. Scale-dependent recolonization: the role of sediment stability in a dynamic sandflat habitat. Ecology, 77: 2472–2487.

Tuck, I.D., Hall, S.J., Robertson, M.R., Armstrong, E., Basford, D.J. 1998. Effects of physical disturbance in a previously unfished sheltered sea loch. Marine Ecology Progress Series, 162: 227–242.

Underwood, A.J. 1997. Experiments in ecology. Cambridge University Press, Cambridge, UK.

Van Dolah, R.F., Wendt, P.H., and Nicholson, N. 1987. Effects of research trawl on a hard bottom assemblage of sponges and corals. Fisheries Research, 5: 39–54.

6.2.4 Long-term trends in demersal fish and benthic invertebrates

To understand the long-term influence of fishing activity on the benthic environment it is necessary to have access to two types of data. The first is samples of benthic organisms collected in a standardised manner over a long period of time. These would preferably represent several trophic levels of benthic invertebrates and fish, from small-bodied infaunal species (e.g., polychaetes, copepods) to larger-bodied macro-invertebrates and fish. In addition, it is necessary to have detailed records of the spatial distribution and fishing effort of the major bottom trawl fleets throughout the area of study, and for a long time period. Unfortunately, for the past 100 years during which fishing in the waters of northwestern Europe has dramatically increased, these comprehensive data are not available. In their absence, we must use data that are available, but which were not collected with this purpose in mind. An understanding of the cause of species trends is fundamental, and can be helped by information on life-history strategy (growth rate, reproductive potential, etc.) or body form (mobility, fragility, etc.). Unfortunately, without a greater understanding of the theoretical framework to explain these ecosystem processes, it is often only possible to show correlations between fishing effort and species trends, without demonstrating an irrefutable causative link.

6.2.4.1 Review of the IMPACT II report

This chapter of the IMPACT II report includes analyses of benthic data sets that cover different periods during the Twentieth Century. They include two sections describing benthic invertebrate data sets, two sections detailing trends in some benthic taxa and some fish taxa, and three describing time series of fish catches. All data sets are from the North

Sea and most refer to the southern North Sea/German Bight. This analysis is followed by a brief summary of other time series data that are relevant and which have been used as additional evidence when preparing recommendations for mitigation. Table 6.2.4.1.1 provides basic information regarding the methodology and data collected in each of the seven studies. The following sections briefly review their main results and conclusions. They also consider any problems with the studies that may affect the extent to which their conclusions can be justified.

6.2.4.1.1 Historical and recent data on epifauna in the southern North Sea (IMPACT II, Chapter 3.8.1)

Although the gears used for sampling on each occasion are known, there are no details of mesh size or duration of hauls, so the data have only been analysed as presence/absence (Table 6.2.4.1.1). The comparability of these sampling gears is identified as one of the main uncertainties. For example, the authors recognise that the decline in catches of small echinoderms (*Echinocyamus pusillus*) and small bivalves (*Abra* and *Nucula*) may be an artefact of sampling. The consistency of these trends in abundance at higher trophic levels suggests that observed trends in species abundance are robust.

The authors identify a decline in the occurrence of bivalves and an increase in scavenging crustaceans (decapods), gastropods (*Buccinum, Lunatia*), and echinoderms (*Astropecten, Asterias*). They attribute these latter changes to the effect of fisheries, which increase productivity of the seabed through increase in discards and physical damage to other benthic species.

There are some concerns with the use of these data to make inferences about time-series trends, particularly in relation to sampling effectiveness. Firstly, the decline in sampling rates during the early period suggests selective sampling, and therefore a possible over-representation of rare species in samples. Secondly, the consistency of abundance trends at higher trophic levels was interpreted as suggesting that invertebrates within the same taxa/phyla respond to fishing effects in the same way. This may not be the case for a diverse group such as the echinoderms, which have a range of life-history strategies (sessile filter feeders, mobile scavengers, etc.) that can influence their vulnerability to trawl effects.

This study therefore shows that bottom trawling can lead to an increase in scavenging species and is more likely to affect fragile species.

6.2.4.1.2 Historical and recent data on macrofauna in the German Bight (IMPACT II, Chapter 3.8.2)

The German Bight has been sampled on three occasions between 1923 and 1995 with a range of grabs and dredges. The types of grabs used are directly comparable as there is no difference in the area covered and in the penetration depth (Table 6.2.4.1.1). There were, however, some differences in the handling of the samples which were sieved over 1 mm mesh screens in all three investigations. During the 1925 survey, only the larger organisms were identified to species level on board ship, and the rest was summarised in higher taxonomic groups. The samples from 1975 and 1995 are directly comparable.

The authors conclude that the spatial distribution of faunal associations in the German Bight have been relatively stable over the last eighty years, during a period of increasing impact on benthos assemblages by fisheries. The number of organisms, the number of species, and the biomass of nearly all species groups in all associations were in 1975 already much higher than the average values from 1923/1924 and increased again from 1975 to 1995. Comparing 1923/1924 with 1975 could be an artefact due to the gear differences already mentioned. Seasonal differences may also play a part. Data for the earlier period were collected in May and June, but in the two later periods, the data were collected in autumn when benthic biomass tends to be higher. Within the separate macrofauna associations, the authors observed a change in community structure, which is considered to be robust, with a dominance of opportunistic short-lived species and a decrease in long-living sessile organisms.

The authors also discussed the reasons for these trends and the difficulties in separating the effect of fisheries from the multiple influences of pollution and eutrophication. They came to the conclusion that fishing activity is responsible for these changes, for they found extremely high numbers of opportunistic species, which can colonise disturbed areas very quickly. The surveyed area is indeed the heaviest fished area by beam trawlers in the North Sea.

This study shows that bottom trawling can lead to changes in species abundance. Populations of species with a high turnover rate increased in abundance, while populations of species with a low turnover rate declined.

Table 6.2.4.1.1. Summaries of long-term trend studies in IMPACT II.

Study No.	Study Type	Periods Covered	Organisms/ Factors Surveyed	Survey Techniques	Sampling Effort	Time of Survey	Area of Survey	Parameters Measured
1	Time period comparison	a)1902–1912 b)1986	Benthos	a) Dredge and different trawls b) Standard dredge and beam trawl	a) 56 stations b) 40 stations	a) Feb. to Nov. b) April to May	Southern North Sea	Species richness and composition. (% occurrence)
2	Time period comparison	a)1923–1924 b)1975 c)1995	Benthos	a) Petersen grab b) Van Veen grab c) Van Veen grab	a) 137 stns. b) 66 stations c) 32 stations	a) May to June b) October c) Sept. to Oct.	German Bight	Species composition clustering. Species richness, abundance and biomass by phylla.
3	Time series trends	1947–1981	Fish and benthos	Fishermen's hand-ins	250 vessels (total)	All year	Southeast North Sea	Species relative abundance.
4	Time series trends	a)1980–1995 b)1969–1990 c)1972–1991 d)1980–1993	a) Fish b) Fish c) Benthos d) Fish	a) Otter trawl b) Beam trawl c) Beam trawl d) Beam trawl	a) 17 stations b) 4 stations c) 10 stations d) 6 stations	a) February b) Sept. to Nov. c) Sept. to Nov. d) Sept. to Nov.	Southeast North Sea	Trends in log-transformed relative abundance.
5	Time series trends	1972–1994	Fish	Stake or fyke net	Monthly average of daily counts	July to Oct.	Wadden Sea	Species recruit abundance and environmental factors.
6	Time series comparison	a)1974–1993 b)1977–1993 c)1977–1993 d)1987–1996	a) Beam effort b) Otter effort c) Pair Trl. effort d) Dab and G. gurnard	a) Log books b) Log books c) Log books d) GOV trawl	a)> German b)> and Dutch c)> fleets d) 20–30 trl.y–^{1}	a) Annual b) Annual c) Annual d) Quarters 2 and 3	German Bight	Effort (hours fished) and species catch rates.
7	Time series trends	a)1962–1995 b)1959–1994	a) Fish (8 species) b) Fish (8 species)	a) Otter trawl b) Otter trawl	a) 5778 trawls b) 2198 trawls	a) Quarter 1 b) Quarter 3	North Sea (by 8 sub-areas)	Species catch rates (total and by size category and area)

6.2.4.1.3 Long-term impact of bottom fisheries on several by-catch species of demersal fish and benthic invertebrates in the southeastern North Sea (IMPACT II, Chapter 3.8.3)

This study modelled the long-term trends in relative abundance of a selection of fish and invertebrates delivered to the Den Helder Zoological Station since 1947 by fishermen. All sorts of factors may have affected hand-in rates, thus this study may have been subject to sampling variation problems. The extent to which this may have affected the results and conclusions cannot readily be assessed. For some species (common skate, greater weever, whelk, lobster), the estimated catchability suggests that populations of these species were reduced to close to zero by the mid-1980s. These data provide some evidence of an association between these species declines and increased fishing effort, although decreased catch rates of *Neptunea* are contradicted by German Poseidon studies from the early years of the Twentieth Century, which showed an increasing trend (Chapter 3.8.1, IMPACT II) (Philippart, 1998).

Although many of the effects are consistent with what we already know about the effects of fishing on vulnerable populations, particularly of elasmobranchs, some declines in abundance, for example, that of the greater weever, do not fit into this pattern and cannot easily be explained by fishing effects alone. These data reaffirm the need to include other factors, especially those which describe long-term environmental change, in order to address these more complex situations.

This study shows that bottom trawling can lead to a change in species abundance and a decrease in the abundance of species with low rates of turnover. However, WGECO has reservations about the strength of the evidence for the conclusions reached in this section.

6.2.4.1.4 Shifts in the benthic community of the southeastern North Sea during extensive bottom trawl fishery (IMPACT II, Chapter 3.8.4)

Research vessel survey data sets are probably the best means of identifying long-term trends in species abundance. The standardised nature of sampling in space and time, with additional information on age and size structure, provides high quality data, even though the longest of such series only extend back to the 1970s. This study used a regression analysis to determine the trends over the time series. They then used the statistical significance of the analysis to determine which were valid trends and which were not. However, because the data were time series, this approach is not valid because the consecutive data are autocorrelated and therefore not independent. The statistical significance of these results must be questioned.

Changes in fish populations were found to be greatest in areas that were moderately to extensively fished. Increases in catch rates of dragonets, dab, grey gurnard and starfish were suggested as being a response to the disturbance caused by fishing, although the authors could not exclude other influences on these populations such as eutrophication or long-term variability in climatic conditions.

Although these survey data are good for identifying robust trends in increased populations of scavenger species and an associated increase in species abundance and distribution, they do not improve our ability to explain their cause.

6.2.4.1.5 Long-term fluctuations in fish recruit abundance in the western Wadden Sea in relation to variation in the marine environment (IMPACT II, Chapter 3.8.5)

Variations in recruit abundance of twelve fish species in the Wadden Sea were compared to four very crude indices of environmental condition, to test the hypothesis that the number of recruits was related to environmental variation. They used the North Atlantic Oscillation (NAO) as an index of eastward transport across the North Sea, and winter temperature, mean annual chlorophyll and crustacean biomass as an indication of environmental conditions within the Wadden Sea. Analysis of the 1974–1994 time series of recruit abundance variation showed a coincidence of these variables for six of the twelve species (herring, flounder, sole, plaice, cod and whiting), corresponding to coherent changes in environmental features. For these species, years with high recruit abundance correspond to strong eastward transport (low NAO) and severe winters (low temperature). Low crustacean biomass and high chlorophyll coincided with high recruit abundance. The authors conclude that the variation in abundance of recruits in the nursery area of the Wadden Sea is influenced by the physical forcing of the environment, although it is unlikely that the same physical process should regulate the recruit abundance of six fish species, at least all in the same way.

Our evaluation of this section concludes the following: a) the study is spatially limited to a unique habitat within the western Wadden Sea (Marsdiep), b) correlations do not confirm causality, and c) it is uncertain whether environmental factors cause long-term trends in populations, or introduce variability around a mean. Furthermore, this study made little attempt to relate changes in fish recruitment, albeit in one small site within the North Sea, to any variation in fishing disturbance. As several of the species were of commercial importance, this would suggest that an analysis of recruitment on a North sea scale may be more appropriate. It is worth adding that any observed changes in recruit abundance at this site may not in fact reflect variation in recruitment in the North Sea generally, but could alternatively indicate changes in distribution, or some combination of the two.

6.2.4.1.6 Abundance of dab and grey gurnard and trawlable biomass in relation to fishing effort (IMPACT II, Chapter 3.8.6)

In a small but heavily fished part of the North Sea (ICES rectangle 37F7) with good time-series data, a correlation was calculated between fishing effort and survey data. In order to calculate the correlation coefficient, CPUE data for dab and grey gurnard were used for the period 1973 to 1994, with Dutch and German fishing effort data. It was expected that, despite the good datasets, the area would be too small to show any significant correlations. Due to the high inter-annual variation both in fishing effort and CPUE data, this expectation was confirmed for these two species.

This analysis confirms that broader-scale studies may be necessary to identify such trends, and so no evidence of decreased population size as a direct result of fishing was apparent.

6.2.4.1.7 Trends in abundance and length of eight target and non-target fish species in the North Sea (IMPACT II, Chapter 3.8.7)

Another series of research survey data, on this occasion for eight species collected throughout the North Sea, was analysed for temporal trends. The collation of these time series data sets involved the aggregation of trawl sample data collected by several nations, using a number of different vessels and several different gears. The possibility that the inclusion of this level of sampling variation could cause some problems in interpreting the results was not discussed. However, we do not consider that this sampling variability has affected the direction of any perceived change, rather that it would have made the detection of any trends more difficult. Once again, this study also used an inappropriate parametric regression analysis to test the statistical significance of time series data. However, the data were provided in considerable detail, allowing us to evaluate their results.

Trends in abundance of all eight species over the whole North Sea were confusing. Anglerfish and starry ray showed clear increasing trends in the Quarter 1 survey data, but these were not so apparent in the Quarter 3 time series. Apart from a high abundance value in 1976, an increase in dab abundance was also indicated by the Quarter 1 data, but once again this trend was not apparent in Quarter 3 survey data sets. The Quarter 1 time series suggested that plaice increased up to the late 1980s and then declined. Once more these trends were not obvious in the Quarter 3 data. The discrepancy between the trends indicated by the two data sets was a consistent feature of this study.

Comparison of spatial patterns in abundance and body-size trends was more informative in some cases, but not in others. For example, the considerable decline in the abundance of spurdog, particularly larger individuals, in the northwestern North Sea, but not elsewhere, was apparent in both data sets. The same trends were apparent for thornback ray, but were not so obvious. Anglerfish catch rates increased markedly in the northwestern North Sea according to the Quarter 1 data, associated with a large increase in the proportion of larger individuals in the catch. Abundance and size change trends in the southern North Sea were not so apparent. These trends in abundance were not apparent in the Quarter 3 data. The Quarter 1 data indicated increases in the abundance of dab in some areas of the North Sea. These areas were not contiguous. In other areas no trend was apparent. Only in the central North Sea was the same increasing trend apparent in Quarter 3. Elsewhere the data were generally too variable to determine with confidence any trends. No real trends in dab body size were apparent. The situation for grey gurnard was very similar to the dab, except that in the northern North Sea a clear trend for decrease in size was apparent. Trends in the abundance of plaice in different parts of the North Sea were not clear, but clear trends towards decreasing body size were indicated in many different areas of the North Sea.

To conclude, this study presented a considerable amount of information, but the summarising of trends was over-simplified and confounded in some respects by the inappropriate regression analysis.

This study shows evidence of increasing and decreasing trends for North Sea fish populations. It is not clear what has caused these trends.

6.2.4.2 Discussion of other long-term studies

Analysis and interpretation of the effects of fishing on the marine benthic environment using time-series data (showing trends in abundance) have been the subject of several major reviews (Jennings and Kaiser, 1998; Daan and Richardson, 1996; Gislason and Sinclair, in press). There are also a number of other analyses of demersal fish and benthic invertebrates data sets which are relevant to the issues discussed here. This section reviews the conclusions provided in the IMPACT II report, together with other relevant supporting text, for a series of generic effects identified by WGECO under the broad headings of species and habitats. Each effect of bottom trawling is described with reference to the evidence here and elsewhere from long-term effects.

6.2.4.2.1 Habitat change

Bottom trawls can remove some physical features

There are few references to the long-term effect of bottom trawling on the degradation of the physical structure of the seabed. On Jeffreys Bank, Gulf of Maine, USA, boulders up to 2 m in diameter were repeatedly moved over six years of fishing. The thin veneer of mud overlaying the gravel beds was also markedly reduced (Auster *et al.*, 1996). There are many references to the direct effects of trawling on physical features and habitats. There will, in theory, be cumulative effects of this trawling activity, but where these are described they refer to the degradation of structural biota (see Section 6.2.4.2.2).

Bottom trawling can cause a reduction in structural biota

Despite limited evidence presented in the IMPACT II report, there is strong evidence from elsewhere that organisms which form part of the biogenic structure in an area can be destroyed or removed by the continuing passage of bottom trawls over many years, e.g., hydroids, bryozoans, sponges, and serpulid worm matrices (Bradstock and Gordon, 1983; Sainsbury, 1987; Collie *et al.*, 1997). Generally, time series are short (years to decades) and do not cover the entire period of fishing activity.

Bottom trawling can cause a reduction in complexity

We can find no evidence to support this conclusion from interpretation of long-term data sets.

Bottom trawling can cause a reduction in the physical structure of the sea floor

We can find no evidence to support this conclusion from interpretation of long-term data sets.

6.2.4.2.2 Species composition

Bottom trawling can cause a reduction in the geographic range of species

There is some evidence from the long-term studies in this chapter of the IMPACT II report to support this conclusion. Evidence is weak largely because long time-series data sets rarely have a spatial component as well, thus limiting this analysis of regional variability. A number of well-known studies from the literature have suggested these range effects, but the possible contribution of other environmental and/or anthropogenic factors cannot easily be discounted. Theoretical studies, and analysis of the elasmobranch stocks in the North Sea, suggest that populations of many species of skates and rays have declined over a long time period, and that these changes also have a spatial aspect. In the western Atlantic, the decline in spatial extent of the barndoor skate has been associated with the effect of fishing (Casey and Myers, 1998). These changes cannot be unambiguously assigned to the effects of bottom trawling. Additional data are reviewed in the next section.

Bottom trawling can cause a decrease in populations that have low rates of turn-over

There are few data in the chapter of the IMPACT II report dealing with long-term change to support this conclusion of the general effects of fishing. Theoretical population dynamics suggest that populations of slow-growing animals, which grow to a large ultimate size, and mature at a late age and large size, should decrease first when subject to an increase in mortality rate (Southwood, 1976; Hoenig and Gruber, 1990). Of 18 intensively exploited fish stocks in the Northeast Atlantic, those which decreased in abundance matured later and at a larger size, attained a larger maximum size, and had faster daily specific growth rates compared with their nearest phylogenetic relatives (Jennings *et al.*, 1998). Jennings *et al.* (1999) have shown that for the whole aggregated assemblage of individuals belonging to all species, mean growth rates and mean ultimate body size declined over the period 1925–1996. This is associated with increased fishing effort. Greenstreet and Rogers (in press) have demonstrated that such trends are apparent when only non-targeted species are included in the analysis.

Elasmobranchs are generally slow growing, mature late, and have lower potential rates of population increase. This makes them vulnerable to exploitation (Hoenig and Gruber, 1990; Jennings *et al.*, 1998). Of all the marine organisms available for study, this group may most clearly identify the effects of fishing activity because of these characteristics. Catches of demersal elasmobranchs caught as a by-catch of mixed fisheries have declined, for example, the common skate in the Irish Sea (Brander, 1981) and the barndoor skate *R. laevis* in the Northwestern Atlantic (Casey and Myers, 1998). In theory, using only knowledge of their life-history characteristics, changes in the relative abundance of sharks and rays in northern European waters should correspond to their perceived level of vulnerability to fishing mortality. The ranking order of a selection of rays, from the most to the least vulnerable, has been suggested as *R. batis* > *R. clavata* > *R. montagui* > *R. naevus* > *R. radiata* (Walker and Hislop, 1998). In practice, records from contemporary surveys tend to support this theoretical vulnerability approach. In the North Sea, while several ray species have shown constant or declining catch rates, only the starry ray *R. radiata*, which is the only one discarded by trawl fisheries, appears to have increased in abundance. Hence, a combination of both discarding and favourable life history traits lead to high abundance. This species also has the smallest ultimate size, the fastest growth rate, and the shortest length and youngest age at maturity (Walker and Hislop, 1998), thus, its life history traits characterize it as the least vulnerable of all skate species. The spotted ray *R. montagui* is considered vulnerable to fishing activity in the southern North Sea, and over-exploitation is thought to have caused an extirpation of these species from the German Bight in the southern North Sea (Walker and Heessen, 1996).

Bottom trawling is patchy and can cause fragmentation of populations

We can find no evidence to support this conclusion from interpretation of long-term data sets.

Fragile species are more affected by bottom trawling than robust species

We can find no evidence to support this conclusion from interpretation of long-term data sets.

Surface-living species are more affected by bottom trawling than deep-burrowing species

We can find no evidence to support this conclusion from interpretation of long-term data sets.

The relative abundance of species is altered by bottom trawling

There is clear evidence from the chapter of the IMPACT II report (Chapters 3.8.1, 3.8.2, and 3.8.4) dealing with long-term data sets that species relative abundance has changed over a long period. Concluding that bottom trawling is the only cause of this variability is a concern that is recognised, but cannot be evaluated further. Analysis of the *Abra – Pectinaria* community of the Channel in the 1980s and 1990s, compared with data from the 1920s, showed that, although some bivalve species showed variable density, the community as a whole was stable and persistent. Several changes in macrofauna abundance were documented between samples collected in the 1950s and 1980s from the Dogger Bank in the North Sea (Kröncke, 1992; Kröncke and Knust, 1995). However, the loss of *Spisula* from parts of the region was interpreted as responses to food availability due to eutrophication and/or climatic fluctuations, rather than attributed to fishing. Between 1971 and 1986, trends in the abundance of benthic infauna at two sites off the Northumberland coast paralleled changes in phytoplankton index, suggesting that the benthic productivity was controlled by organic matter input (Buchanan, 1993). From 1986 until 1990 there was an increase in fishing effort within the *Nephrops norvegicus* (Dublin Bay prawn) ground which caused a decline in benthic productivity and large-scale year-to-year changes in the relative abundance of different species at the site in the fished area. However, at a site outside the *Nephrops* ground, the benthos community continued to closely track changes in phytoplankton (Frid and Clark, in press). Frid *et al.* (in press) examined macrofauna abundance data in five regions of the central and southern North Sea. Clear patterns of change in the macrofaunal communities were observed in three areas between the early 1920s and the late 1980s. The lack of change in the remaining two areas was interpreted as evidence that the observed patterns were not part of a broad-scale environmental change. One of the areas showing no significant change included much of the Dogger Bank, and this may reflect the long history of fishing activity in this region, which may already have undergone significant change prior to the 1920s. In the three regions that did show significant shifts in community composition, the changes appeared to be the result of altered abundance of several taxa, not just those sensitive to the direct effects of fishing. This suggests that indirect effects such as changes in sediment structure, nutrient flux, predation pressure, etc., all indirect effects of fishing, may be at least as important as the direct effects.

Long-term variation in fish species composition, leading to change in species diversity, has been demonstrated in the northwestern North Sea (Greenstreet and Rogers, in press; Pope *et al.*, 1988). Greenstreet and Rogers (in press) have reviewed the literature describing long-term trends in the numbers of many non-target fish species in the North Sea. Their observations are summarised in Tables 6.2.4.2.1 and 6.2.4.2.2. Many of these same studies were briefly reviewed at the 1997 WGECO meeting (ICES, 1998). Most fish communities that have been studied for a decade or more show changes in the relative abundance of the species present (Jennings and Kaiser, 1998). In a community off Jamaica, the larger targeted fish species disappeared, but there were few changes in non-target species (Koslow *et al.*, 1988). In Thailand large target species of fish declined and were replaced by flatfish, squids and crustaceans in catches (Pauly, 1979). The degree to which this can be attributed to the effect of fishing is unclear.

Additional information has recently been generated by the inclusion of early survey data (1966–1982) from the International Bottom Trawl Survey (IBTS) (Anon., 1999). The average catch as number per hour fished was calculated for all target and non-target fish species distinguished in the database. For ten selected non-target species, the average catch in number was differentiated into three size classes.

Several non-target fish species (e.g., starry ray, four-bearded rockling, bull-rout) in the North Sea have shown an increase in abundance since 1965. The spurdog (*Squalus acanthias*) had a period of very high catches in the late 1970s and has been back to a low level since the early 1980s.

Bottom trawling can have sub-lethal effects on individuals

Trawling has been observed to damage the shells of the bivalve *A. islandica* (Arntz and Weber, 1970; Rumohr and Krost, 1991). Individuals that are only slightly damaged can repair cracks in their shells, and sand grains become lodged in the shell matrix aiding their identification. In this way, Witbaard and Klein (1994) were able to study annual growth of the bivalves and could correlate increased shell damage with increased beam trawling activity between 1972 and 1991.

Table 6.2.4.2.1. Direction of trends in the abundance of non-target species published in recent studies.

Species	**Study**	**Trend in abundance**
Lesser spotted dogfish (*Scyliorhinus canicula*)	Heessen (1996)	Not determinable
Tope (*Galeorhinus galeus*)	Heessen (1996)	Possible increase
Smooth hound (*Mustelus mustelus*)	Heessen (1996)	Possible increase
Blond Ray (*Raja brachyura*)	Walker and Heessen (1996)	No trend
Spotted Ray (*Raja montagui*)	Rogers and Millner (1996)	Decrease
	Walker and Heessen (1996)	Not determinable
Cuckoo ray (*Raja naevus*)	Walker and Heessen (1996)	Possible decrease
Starry ray (*Raja radiata*)	Heessen and Daan (1996)	Increase
	Walker and Heessen (1996)	Increase
Tusk (*Brosme brosme*)	Heessen (1996)	Possible increase
Five-bearded rockling (*Ciliata mustela*)	Rijnsdorp *et al*. (1996b)	Decrease
Three-bearded rockling (*Gaidropsarus vulgaris*)	Rijnsdorp *et al*. (1996b)	Decrease
Four-bearded rockling (*Enchelyopus cimbrius*)	Heessen and Daan (1996)	Possible increase
Ling (*Molva molva*)	Heessen (1996)	Not determinable
Pollack (*Pollachius pollachius*)	Heessen (1996)	Not determinable
Bib (*Trisopterus luscus*)	Heessen and Daan (1996)	Increase
	Corten and van de Kamp (1996)	No trend
Poor cod (*Trisopterus minutus*)	Rijnsdorp *et al*. (1996b)	Decrease
	Heessen and Daan (1996)	Increase
	Greenstreet and Hall (1996)	No trend
	Corten and van de Kamp (1996)	Increase
Hake (*Merluccius merluccius*)^1	Heessen (1996)	No trend
John Dory (*Zeus faber*)	Heessen (1996)	Possible increase
	Corten and van de Kamp (1996)	Not determinable
Boarfish (*Capros aper*)	Heessen (1996)	Increase
Bluemouth (*Helicolenus dactylopterus*)	Heessen (1996)	Increase
Norway haddock (*Sebastes viviparus*)	Greenstreet and Hall (1996)	Increase
Red gurnard (*Aspitrigla cuculus*)	Rijnsdorp *et al*. (1996b)	Decrease
	Heessen (1996)	Possible increase
Grey gurnard (*Eutrigla gurnardus*)	Rijnsdorp *et al*. (1996b)	Decrease
	Heessen and Daan (1996)	Increase
	Greenstreet and Hall (1996)	Decrease
Tub gurnard (*Trigla lucerna*)	Rijnsdorp *et al*. (1996b)	Possible decrease
	Heessen (1996)	No trend
	Corten and van de Kamp (1996)	Not determinable
Bullrout (*Myoxocephalus scorpius*)	Rijnsdorp *et al*. (1996b)	Possible decrease
	Heessen and Daan (1996)	Increase
Hooknose (*Agonus cataphractus*)	Rijnsdorp *et al*. (1996b)	Decrease
	Rogers and Millner (1996)	No trend
Lumpsucker (*Cyclopterus lumpus*)	Heessen (1996)	Possible increase
Sea snail (*Liparis liparis*)	Rogers and Millner (1996)	No trend
Red mullet (*Mullus surmuletus*)	Rijnsdorp *et al*. (1996b)	Increase

Species	Study	Trend in abundance
	Heessen (1996)	Not determinable
	Corten and van de Kamp (1996)	Not determinable
Ballan wrasse (*Labrus bergylta*)	Rogers and Millner (1996)	Not determinable
Eelpout (*Zoarces viviparus*)	Rogers and Millner (1996)	Decrease
Butterfish (*Pholis gunellus*)	Rogers and Millner (1996)	Decrease
Catfish *(Anarhichas lupus*)	Heessen (1996)	Increase
Lesser weever (*Echiichthys vipera*)	Rijnsdorp *et al*. (1996b)	Decrease
	Heessen (1996)	Increase
	Rogers and Millner (1996)	No trend
Greater weever (*Trachinus draco*)	Rijnsdorp *et al*. (1996b)	Decrease
Dragonet (*Callionymus lyra*)	Rijnsdorp *et al*. (1996b)	Decrease
	Heessen (1996)	Decrease
Spotted dragonet (*Callionymus maculatus*)	Heessen (1996)	Not determinable
Norwegian topknot (*Phrynorhombus norvegicus*)	Heessen (1996)	Possible increase
Scaldfish (*Arnoglossus laterna*)	Rijnsdorp *et al*. (1996b)	Decrease
	Heessen (1996)	Possible increase
Witch (*Glyptocephalus cynoglossus*)	Heessen (1996)	No trend
	Greenstreet *et al*. (in press a)	Decrease[2]
Long rough dab (*Hippoglossoides platessoides*)	Rijnsdorp *et al*. (1996b)	Decrease
	Heessen and Daan (1996)	Increase
	Greenstreet and Hall (1996)	No trend
Halibut (*Hippoglossus hippoglossus*)[1]	Heessen (1996)	Possible increase
Common dab (*Limanda limanda*)	Rijnsdorp *et al*. (1996b)	Decrease
	Heessen and Daan (1996)	Possible increase
	Greenstreet and Hall (1996)	No Trend
Lemon sole (*Microstomus kitt*)[1]	Rijnsdorp *et al*. (1996b)	Possible decrease
	Heessen and Daan (1996)	Increase
	Greenstreet and Hall (1996)	Increase
Solenette (*Buglossidium luteum*)	Rijnsdorp *et al*. (1996b)	Decrease
	Rogers and Millner (1996)	No trend
	Heessen (1996)	Not determinable

[1] This study dealt exclusively with the North Sea where hake, halibut and lemon sole are not targeted specifically, although they are usually landed if taken in the by-catch. In other parts of the continental shelf waters of northern Europe, i.e., in western and southern areas, hake is specifically targeted by fishing operations.

[2] A decrease is suggested in all four of the regions studied.

Table 6.2.4.2.2. Type of study, spatial and temporal coverage, and gears used in the studies cited in Table 6.2.4.2.1.

Study	Study type	Period(s)	Gears	Spatial coverage
Rijnsdorp *et al*. (1996b)	Two-period comparison	1906–1909, 1990–1995	OT20[1] v GOV[2]	Southern North Sea
Heessen and Daan (1996)	Time series	1970–1993	GOV[2,3]	North Sea
Heessen (1996)	Time series	1970–1993	GOV[2,3]	North Sea
Walker and Heessen (1996)	Time series	1970–1993	GOV[2,3]	North Sea
Rogers and Millner (1996)	Time series	1973–1995	BT2 and PN1.5[4]	S.E. English coast
Corten and van de Kamp (1996)	Time series	1970–1993	GOV[2,3]	Southern North Sea
Greenstreet and Hall (1996)	Two-period comparison	1929–1953, 1980–1993	OT48 v AOT[5]	Northwestern North Sea
Greenstreet *et al*. (in press a)	Time series	1925–1996	OT48 v AOT[5]	Northwestern North Sea

[1] 20 ft (6 m) otter trawl with 40 mm codend mesh with a swept area rate of 15 000 m^2 h^{-1}.

[2] Grande ouverture verticale (GOV) otter trawl with 20 mm codend mesh with a swept area rate of 530 000 m^2 h^{-1}.

[3] In the early 1980s countries participating in this ICES coordinated survey switched to using the GOV, prior to this other gears were used, e.g., the herring bottom trawl with 20 mm mesh codend on Scottish vessels.

[4] 2 m beam trawl and 1.5 m push net designed to have similar efficiency and selectivity enabling catches per unit swept area to be compared directly.

[5] These two gears are essentially the same, being 48 ft otter trawls with 35 mm codend mesh.

Bottom trawling can cause an increase in populations that have high rates of turnover

Changes in species composition which broadly support a dominance of opportunistic short-lived species, and a decrease in long-living sessile organisms, are suggested by studies in the German Bight (IMPACT II, Chapter 3.8.2). Even though there is no irrefutable evidence that trawling alone has caused these effects, the weight of evidence in this report and elsewhere, tends to support the validity of their conclusions. There is evidence from studies of demersal fish populations that also suggest that populations of small-bodied fish have increased over a long time period, and that these trends have correlated with changes in fishing effort. Populations of species with a high turn-over rate have increased over several decades in the northwestern North Sea (Greenstreet and Hall, 1996; Greenstreet *et al.*, 1999b), leading to overall trends in the means of many of these characteristics for the assemblage as a whole (Jennings *et al.*, 1999). This has been noted for the whole demersal fish assemblage, as well as for the non-target groundfish assemblage only (Greenstreet and Rogers, in press; Rijnsdorp, *et al.*, 1996b). Although these observations have been noted for fish populations over the long term, to our knowledge this effect has not been demonstrated in the benthos.

Bottom trawling favours populations of scavenging species

There are no specific long time series studies of scavengers in the IMPACT II report, and we know of no other data from the literature for benthos. The effect is however well documented for other parts of the ecosystem. For example, analysis of the meroplankton recorded in the North Sea by the Continuous Plankton Recorder (CPR) surveys (Warner and Hays, 1994) have shown an increase in the dominance of echinoderm larvae (Lindley *et al.*, 1995). It has been suggested that this reflects an increase in the relative abundance of scavenging species such as sea stars and ophioroids following fishing disturbance. However, the ability of the CPR programme to quantify plankton and especially meroplankton is limited and the timing of the change does not coincide with the major changes in North Sea fishing activity. This suggests that environmental factors may be at least partly responsible for the patterns in plankton abundance observed in the CPR data. It is doubtful if further analysis of these data will contribute to the analysis of macrobenthic scavenger species. The increase in certain scavenging seabirds as a direct result of fishing activities is well documented (ICES, 1996).

6.2.4.3 Other potential causative factors

Changes in fishing practices are far from being the only changes to have occurred in the North Sea during the last few decades. Variation in sea surface temperature and salinity, the strength of Atlantic inflow in the north, and dissolved inorganic phosphate are just some of the factors for which decadal scale changes have been documented (Becker and Pauly, 1996; Danielssen *et al.*, 1996; Laane *et al.*, 1996; van Leussen *et al.*, 1996; Turrell *et al.*, 1996). Similarly, chlorophyll concentrations, primary productivity, and phytoplankton, zooplankton and benthos species composition and abundance are just a few components of the biota for which long-term variation has been described (Reid *et al.*, 1990; Beukema *et al.*, 1996; Bot and Colijn, 1996; Frid and Huliselan, 1996; Greve *et al.*, 1996; Frometin and Planque, 1996; Frometin *et al.*, 1997a, 1997b; Planque and Taylor, 1998). All these variables, and probably many more, have affected the population dynamics of fish communities (e.g., Nielsen and Richardson, 1996; Rogers and Millner, 1996; Corten and van de Kamp, 1996; Heessen and Daan, 1996).

Furthermore, although it is generally assumed that fishing disturbance in the North Sea has increased over the course of the Twentieth Century, the development of the fishing industry here has been far from simple (ICES, 1995). Long-term changes in fishing effort reveal a complex pattern of spatial and temporal interactions (Greenstreet *et al.*, 1999a; Jennings *et al.*, 1999). Thus, the changes in the pattern of fishing effort distribution over the past two to three decades may perhaps be potentially complicated enough to have caused the variety of trends in species abundance shown in Table 6.2.4.2.1. Few species groups show consistent trends in abundance, and several were contradictory. These data confirm the complex relationships between biotic and abiotic factors in the marine environment. Designing an experiment capable of teasing out these relationships is complex.

6.2.4.4 Limitations of time series data studies

Despite the importance of time series (or long-term) studies, many problems are frequently encountered with the use and analysis of these data. These include the problems of finding data covering long periods that are consistent in the species recorded or that use constant sampling techniques.

The short duration of many data sets makes difficult the identification of fishing as a causative factor determining changes. In most ecosystems, even the longest time series started well after the initiation of fishing activities (Cushing 1988; Smith 1994). Thus, these data do not necessarily reflect the changes between an unfished situation and a fished one. Furthermore, regular standardised surveys are fairly recent in all areas, so data time series are relatively short.

Some data sets which predate the 1920s have been identified (Ruud 1968; Johannessen and Sollie, 1994; Greenstreet and Hall, 1996; Quero and Cendrero, 1996; Rogers and Ellis, 2000), and some groundfish survey data from the early 1900s also exist (e.g., Rijnsdorp *et al.*, 1996a), but data from the 18th or 19th centuries are scarce (Quero, 1998). The longest time series of fish abundance covering the whole North Sea are those from the winter International Bottom Trawl Survey (IBTS, initially the International Young Fish Survey, IYFS) which only extends back to 1970 (ICES, 1999). In other areas, such long time series are not available, for example, in the southern European Atlantic (ICES Divisions VIIIc and IXa) where standardised routine bottom trawl surveys have been collected since 1979 (Cardador *et al.,* 1997).

Species trends vary according to the time and geographical scale at which they were collected. The time span covered by these time series is important in determining which part of the trend we observe (increase, decrease or stable situation). For example, poor cod in the southern North Sea appear to have increased in abundance over the period 1970 to 1993. However, data spanning a longer period suggest that they decreased in abundance between 1906–1909 and 1990–1995. Distinguishing short-term fluctuations from long-term trends becomes increasingly more problematic as the time span of any data set decreases, a point illustrated by Corten and van de Kamp (1996).

The geographical range covered by the data may also affect how we perceive the trends. Species trends that appear important at a small scale may in fact be less so at a larger scale. Examples of this include large-scale changes in bib (*Trisopterus luscus*) and John Dory (*Zeus faber*) in the North Sea that may not have been observed in studies concentrating on more restricted regions

Other problems have been caused by incomplete recording. Early surveys generally considered only commercially important species (e.g., Jones and Hislop 1978), and a few more abundant non-target species (Richards *et al.,* 1978; Hempel 1978). For example, of the 175 groundfish species recorded in the North Sea (Yang, 1982), fewer than a dozen were included in initial analyses. Only in recent years have trends in the abundance of 44 of the more commonly caught non-target species been studied.

Due to variability in recruitment, immigration and emigration, and statistical difficulties with occurrences of rare events, it is very difficult to confirm the reliability of the abundance indices of rare non-target species. Within the IBTS data set there is one event which helps to verify these indices. Heessen *et al.,* (1996) described the occurrence of the deep-water species blue-mouth rosefish (*Helicolenus dactylopterus*) after a limited number of larvae or very young fish entered the North Sea in 1990/1991. During the following eight years, individuals were widely distributed and were caught throughout the North Sea during the first quarter IBTS. The trend in abundance resembled a typical catch curve with high mortalities in the first and second year and a slower decrease in older ages (Anon., 1999). This is a good example showing that the survey results and the method of calculating the abundance index can give representative results related to rare and widely distributed non-target species, at least under some circumstances.

6.2.4.5 References

Anon. 1999. Final report EU contract no. 95/69: ‘Input of Historic IBTS data’. RIVO-DLO report, CO14/99.

Arntz, W.E., and Weber, W. 1970. *Cyrina islandica* (Mollusca: Bivalvia) als Nahrung von Dorsch und Kliesche in der Kieler Bucht. Berichte der Deutschen Wissenschaftlichen Kommision fur Meeresforschung, 21: 193–209.

Auster, P.J., Malatesta, R.J., Langton, R.W., Watling, L., Valentine, P.C., Donaldson, C.L., Langton, E.W., Shepard, A.N., and Babb, I.G. 1996. The impact of mobile fishing gear on seafloor habitats in the Gulf Maine (Northwest Atlantic): implications for conservation of fish populations. Reviews in Fisheries Science, 4: 185–202.

Becker, G.A., and Pauly, M. 1996. Sea surface temperature changes in the North Sea and their causes. ICES Journal of Marine Science, 53: 887–898.

Beukema, J.J., Essink, K., and Michaelis, H. 1996. The geographical scale of synchronized fluctuation patterns in zoobenthos populations as a key to underlying factors: climatic or man-induced. ICES Journal of Marine Science, 53: 964–971.

Bot, P.V.M., and Colijn, F. 1996. A method for estimating primary production from chlorophyll concentrations with results showing trends in the Irish Sea and Dutch coastal zone. ICES Journal of Marine Science, 53: 945–950.

Bradstock, R.M., and Gordon, D.P. 1983. Coral-like bryozoan growths in Tasman Bay, and their protection to conserve local fish stocks. New Zealand Journal of Marine and Freshwater Research, 17: 159–163.

Brand, A.R., Allison, E.H., and Murphy, E.J. 1991. North Irish Sea scallop fisheries: a review of changes. *In* An international compendium of scallop biology and culture, pp. 204–218. Ed. by S.E. Shumway and P.A Sander. World Aquaculture Society. Baton Rouge.

Brander, K. 1981. Disappearance of common skate *Raja batis* from the Irish Sea. Nature, 290: 48–49.

Buchanan, J.B. 1993. Evidence of benthic pelagic coupling at a station off the Northumberland coast. Journal of Experimental Marine Biology and Ecology, 172: 1–10.

Cardador, F., Sanchez, F., Pereiro, F., Borges, M.F., Caramelo, A., Azevedo, M., and Fernandez, A. 1997. Groundfish surveys in the Atlantic Iberian waters (ICES division VIIIc and IXa) History and perspectives. ICES CM 1997/Y:08. 25 pp.

Casey, J.M., and Myers, R.A. 1998. Near extinction of a large, widely distributed fish. Science, 281: 690–692.

Collie, J.S., Escanero, G.A., and Valentine, P.C. 1997. Effects of bottom fishing on the benthic megafauna of Georges Bank. Marine Ecology Progress Series, 155: 159–172.

Corten, A., and van de Kamp, G. 1996. Variation in the abundance of southern fish species in the southern North Sea in relation to hydrography and wind. ICES Journal of Marine Science, 53: 1113–1119.

Cushing, D.H. 1988. The Provident Sea. Cambridge University Press, Cambridge, UK.

Daan, N., and Richardson, K. (Eds.) 1996. Changes in the North Sea ecosystem and their causes: Århus 1995 revisited. ICES Journal of Marine Science, 53: 879–1225.

Danielssen, D.S., Svendsen, E., and Ostrowski, M. 1996. Long-term hydrographic variation in the Skagerrak based on the section Torungen-Hirtshals. ICES Journal of Marine Science, 53: 917–925.

Frid, C.L.J., and Clark, R.A. 1999. Longterm changes on North Sea benthos: discerning the role of fisheries. *In* Effects of fishing on non-target species and habitats: biological conservation and socio-economic issues. Ed. by M. J. Kaiser and S. J. de Groot. Blackwell Science, Oxford: Blackwell Science. 416 pp.

Frid, C.L.J., Harwood, K.G., Hall, S.J., and Hall, J.A. In press. Long-term changes in the benthic communities on North Sea fishing grounds. ICES Journal of Marine Science.

Frid, C.L.J., and Huliselan, N.V. 1996. Far-field control of long-term changes in Northumberland NW North Sea coastal zooplankton. ICES Journal of Marine Science, 53: 972–977.

Frometin, J.M., Dauvin, J.C., Ibanez, F., Dewarumez, J.M., and Elkaim, B. 1997a. Long-term changes of four macrobenthic community structures. Oceanologica Acta, 20: 43–53.

Frometin, J.M., Ibanez, F., Dauvin, J.C., Dewarumez, J.M., and Elkaim, B. 1997b. Long-term variability of macrobenthic communities on the North-west French coasts. Journal of the Marine Biological Association UK, 77: 287–310.

Frometin, J.M., and Planque, B. 1996. Calanus and environment in the eastern North Atlantic. 2) Role of the North Atlantic Oscillation on *Calanus finmarchicus* and *C. helgolandicus*. Marine Ecology Progress Series, 134: 111–118.

Gislason, H., and Sinclair, M. (Eds.) In press. Symposium on Ecosystem Effects of Fishing. ICES Journal of Marine Science.

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

Greenstreet, S.P.R., and Rogers, S.I. 1999. The effects of fishing on non-target fish species. *In* Effects of fishing on non-target species and habitats: biological conservation and socio-economic issues. Ed. by M. J. Kaiser and S. J. de Groot. Blackwell Science, Oxford. 416 pp.

Greenstreet, S.P.R., Spence, F.E., Shanks, A.M., and McMillan, J.A. 1999a. Fishing effects in northeast Atlantic shelf seas: patterns in fishing effort, diversity and community structure. V. Changes in structure of the North Sea groundfish species assemblage between 1925 and 1996. Fisheries Research, 40 (2): 153–183.

Greenstreet, S.P.R., Spence, F.E., and McMillan, J.A. 1999b. Fishing effects in northeast Atlantic shelf seas: patterns in fishing effort, diversity and community structure. II. Trends in fishing effort in the North Sea by UK registered vessels landing in Scotland. Fisheries Research, 40 (2): 107–124.

Greve, W., Reiners, F., and Nast, J. 1996. Biocoenotic changes of the zooplankton in the German Bight: possible effects of eutrophication and climate. ICES Journal of Marine Science, 53: 951–956.

Hall, S.J., and Harding, M.J.C. 1997. Physical disturbance and marine benthic communities: the effects of mechanical harvesting of cockles on non-target benthic infauna. Journal of Applied Ecology.

Heesen, H.J.L. 1996. Time-series data for a selection of forty fish species caught during the International Bottom Trawl Survey. ICES Journal of Marine Science, 53: 1079–1084.

Heessen, H.J.L., and Daan, N. 1996. Long-term trends in non-target North Sea fish species. ICES Journal of Marine Science, 53: 1063–1078.

Heessen, H.J.L., Hislop, J.R.G., and Boon, T.W. 1996. An invasion of the North Sea by blue-mouth, *Helicolenus dactylopterus* (Pices, Scorpaenidae). ICES Journal of Marine Science, 53: 874–877.

Hempel, G. 1978. North Sea fisheries and, fish stocks—a review of recent changes. Rapports et Procès-verbaux des Réunions Conseil International pour l’Exploration de la Mer, 173: 145–167.

Hoenig, J.M., and Gruber, S.H. 1990. Life history patterns in the elasmobranchs: Implications for fisheries management. *In* Advances in the biology, ecology, systematics, and the status of the fisheries, Vol. 90, pp. 1–16. Ed. by H. L. Pratt, S. H. Gruber, and T. Taniuchi. NOAA Technical Report.

ICES. 1995. Report of the Study Group on Ecosystem Effects of Fishing Activities. ICES Cooperative Research Report, 200. 120 pp.

ICES. 1996. Seabird/fish interactions with particular reference to seabirds in the North Sea. ICES Cooperative Research Report, 216. 87 pp.

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

ICES. 1999. Report of the International Bottom Trawl Survey Working Group. ICES CM 1999/D:2.

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

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

Jennings, S., and Kaiser, M.J. 1998. The effects of fishing on marine ecosystems. Advances in Marine Biology, 34: 201–352.

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

Johannessen, T., and Sollie, A. 1994. Overvåking av Gruntvannsfauna på Skagerrakkysten. Fisken og Havet, 10: 1–82.

Jones, R., and Hislop, J.R.G. 1978. Changes in North Sea haddock and whiting. Rapports et Procès-verbaux des Réunions Conseil International pour l'Exploration de la Mer, 172: 58–71.

Koslow, J.A., Hanley, F., and Wickland, R. 1988. Effects of fishing on reef fish communities at Pedro Bank and Port Royal Cays, Jamaica. Marine Ecology Progress Series, 43: 201–212.

Kröncke, I. 1992. Macrofauna standing stock of the Dogger Bank. A comparison. 3. 1950–1954 versus 1985–1987—a final summary. Helgolander Meeresuntersuchungen, 46: 137–169.

Kröncke, I., and Knust, R. 1995. The Dogger Bank—A special ecological region in the central North Sea. Helgolander Meeresuntersuchungen, 49: 335–353.

Laane, R.W.P.M., Southward, A.J., Slinn, D.J., Allen, J., Groeneveld, G., and de Vries, A. 1996. Changes and causes of variability in salinity and dissolved inorganic phosphate in the Irish Sea, English Channel and Dutch coastal zone. ICES Journal of Marine Science, 53: 933–944.

Lindley, J., Gamble, J., and Hunt, H. 1995. A change in the zooplankton of the central North Sea 55º to 58º N: a possible consequence of changes in the benthos. Marine Ecology Progress Series, 119: 299–303.

Nielsen, E., and Richardson, K. 1996. Can changes in the fisheries yield in the Kattegat 1950–1992 be linked to changes in primary production. ICES Journal of Marine Science, 53: 988–994.

Pauly, D. 1979. Theory and management of tropical multispecies stocks: a review with emphasis on the Southeast Asian demersal fisheries. ICLARM Studies and Reviews, 1: 1–35.

Philippart, C.J.M. 1998. Long-term impact of bottom fisheries on several bycatch species of demersal fish and benthic invertebrates in the southeastern North Sea. ICES Journal of Marine Science, 55: 342–352.

Planque, B., and Taylor, A.H. 1998. Long-term changes in zooplankton and the climate of the North Atlantic. ICES Journal of Marine Science, 55: 644–654.

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

Quero, J.-C. 1998. Changes in the Euro-Atlantic fish species composition resulting from fishing and ocean warming. Italian Journal of Zoology, 65 (Supplement): 493–499.

Quero, J.-C., and Cendrero, O. 1996. Incidence de la pêche sur la biodiversité ichtyologique marine: Le bassin d' Arcachon et le plateau continental sud Gascogne. Cybium, 20: 323–356.

Reid, P.C., Lancelot, C., Gieskes, W.W.C., Hagmeier, E., and Weichart, G. 1990. Phytoplankton of the North Sea and its dynamics: a review. Netherlands Journal of Sea Research, 26: 295–331.

Richards, J., Armstrong, D.W., Hislop, J.R.G., Jermyn, A.S., and Nicholson, M.D. 1978. Trends in Scottish research-vessel catches of various fish species in the North Sea. Rapports et Procès-verbaux des Réunions Conseil International pour l'Exploration de la Mer, 172: 211–224.

Rijnsdorp, A.D., Buijs, A.M., Storbeck, F., and Visser, E. 1996a. Micro-scale distribution of beam trawl effort in the southern North Sea between 1993 and 1996 in relation to the trawling frequency of the sea bed and the impact on benthic organisms. ICES Mini-symposium on 'Ecosystem effects of fisheries.' ICES CM 1996/Mini:11.

Rijnsdorp, A.D., and van Leeuwen, P.I. 1992. Density-dependent and independent changes in somatic growth of female North Sea plaice *Pleuronectes platessa* between 1930 and 1985 as revealed by back-calculation of otoliths. Marine Ecology Progress Series, 88: 19–32.

Rijnsdorp, A.D., van Leeuwen, P.I., Daan, N., and Heessen, H.J.L. 1996b. Changes in the abundance of demersal fish species in the North Sea between 1906–1909 and 1990–1995. ICES Journal of Marine Science, 53: 1054–1062.

Rogers, S.I., and Ellis, J.R. In press. Changes in the demersal fish assemblages of British coastal waters during the 20th century. ICES Journal of Marine Science.

Rogers, S.I., and Millner, R.S. 1996. Factors affecting the annual abundance and regional distribution of English inshore demersal fish populations: 1973 to 1995. ICES Journal of Marine Science, 53: 1094–1112.

Rumohr, H., and Krost, P. 1991. Experimental evidence of damage to the benthos by bottom trawling with special reference to *Arctica islandica*. Meeresforschung, 33: 340–345.

Ruud, J.T. 1968. Changes since the turn of the century in the fish fauna and the fisheries of the Oslofjord. Helgoländer Meeresuntersuchungen, 17: 510–517.

Sainsbury. 1987.

Smith, T.D. 1994. Scaling Fisheries: The Science of Measuring the Effects of Fishing, 1855–1955. Cambridge University Press, Cambridge.

Southwood, T.R.E. 1976. Bionomic strategies and population parameters. *In*: Theoretical Ecology: Principles and Applications, pp 26–48. Ed. by R.M. May. Blackwell Scientific Publications, Oxford.

Turrell, W.R., Slesser, G., Payne, R., Adams, R.D., and Gillibrand, P.A. 1996. Hydrography of the East Shetland Basin in relation to decadal North Sea variability. ICES Journal of Marine Science, 53: 899–916.

van Leussen *et al*. 1996.

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

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

Warner, A.J, and Hays, G.C. 1994. Sampling by the continuous plankton recorder survey. Progress in Oceanography, 34: 237–256.

Witbaard, R., and Klein, R. 1994. Long-term trends on the effects of the southern North Sea beam trawl fishery on the bivalve mollusc *Arctica islandica*. ICES Journal of Marine Science, 51: 99–105.

Wood, H. 1956. Fisheries of the United Kingdom. *In* Sea Fisheries: Their investigation in the United Kingdom. Ed. by. M. Graham. Arnold, London.

Yang, J. 1982. An estimate of the fish biomass in the North Sea. Journal du Conseil International pour l’Exploration de la Mer, 40: 161–172.

6.2.5 Summary

The information from all the studies in IMPACT II and the additional literature reviewed by WGECO are summarized in Table 6.2.5.1. This table shows which types of studies provide evidence for which types of effects.

Table 6.2.5.1. Summary of effects supported by examples from either IMPACT III or other studies presented in Section 6.2. ʻYES' indicates that evidence was found in either of the two sources; 'NO' does not mean that these effects do not exist, but rather, no evidence was found within this review; N/A indicates that these effects are not applicable to this study type.

	6.2.1. Gears, fleets and physical impacts	**6.2.2. Direct mortality**	**6.2.3. Comparison of disturbed and undisturbed areas**	**6.2.4. Long-term studies**
A) HABITATS				
Bottom trawls can remove some physical features.	YES	N/A	YES	YES
Bottom trawling can cause a reduction in structural biota.	YES	N/A	YES	YES
Bottom trawling can cause a reduction in complexity.	YES	N/A	YES	NO
Bottom trawling can cause a reduction in the physical structure of the sea floor.	YES	N/A	YES	NO
B) SPECIES				
Bottom trawling can cause a reduction in the geographical range of a species.	N/A	N/A	NO	YES
Bottom trawling can cause a decrease in populations which have low rates of turnover.	N/A	NO	YES	YES
Bottom trawling is patchy and can cause fragmentation of populations.	N/A	N/A	YES	NO
The relative abundance of species is altered by bottom trawling.	N/A	YES	YES	YES
Fragile species are more affected by bottom trawling than robust species.	N/A	YES	YES	NO
Surface-living species are more affected by bottom trawling than deep-burrowing species.	N/A	YES	NO	NO
Bottom trawling can have sub-lethal effects on individuals.	N/A	YES	YES	YES
Bottom trawling can cause an increase in populations which have high rates of turnover.	N/A	NO	NO	YES
Bottom trawling favours populations of scavenging species.	N/A	YES	YES	YES

6.3 Potential Effects of Bottom Trawls

6.3.1 Introduction

This section considers the effects of bottom trawls (beam and otter trawls) on habitats and species in the shelf seas of northwestern Europe, and as reviewed in the previous sections of this report (IMPACT II and other literature). Although effects will be specific to gear and area, we are confident that classes of effects are highly applicable to these areas and elsewhere. We recognise that other types of towed bottom gears, such as dredges, can sometimes cause severe effects on the seabed. However, these gears are outside the present terms of reference and have been excluded from the analysis which follows.

It must be recognised that any classification of effects is to an extent arbitrary, with the location of classification boundaries between types of effects depending to some extent on the interests of those proposing the categories. It is also important that in any given situation many or all of the effects will be operating together, and will interact. Such interactions may be additive and may also act synergistically. In addition, trawling is only one of a number of processes operating simultaneously which together cause change in marine systems. These include both anthropogenic changes (eutrophication, pollution, other fishing gear) and processes outside the immediate control of humans (long-term climate change and short-term weather events). Distinguishing among various possible causes of change in marine populations, given the range of possible influences, has proved extremely difficult, even with the best available data sets (Daan and Richardson, 1996).

The studies reviewed in Section 6.2, above, not only provide an inventory of specific effects on benthic habitats and relatively large species, but they also highlight a number of generic issues. These are:

- **Low energy environments are more affected by bottom trawling.** Environments that are frequently disturbed by natural processes (storms, tidal scour, etc.) are little affected, or may even be unaffected, by bottom trawling. Conversely, benthic species assemblages in low energy environments are more affected by bottom trawling. As a general principle, those communities which occupy consolidated sediments (mud, gravel, boulders) are more vulnerable to fishing effects than communities inhabiting unconsolidated sediments (mobile sand) that are frequently resuspended by natural perturbations.
- **Bottom trawling can affect the potential for habitat recovery.** The impact of bottom trawling on marine ecosystems and the seabed may permanently compromise their ability to return to their original condition. Therefore, the cessation of trawling may not always result in a return to the original pre-impacted state. Even untrawled areas are likely to change over time, so the whole concept of ‘original pre-impacted state’ could be challenged. Nonetheless, the dynamics of natural processes are not all the same as the effects of fishing, and the absence of stable natural equilibria in marine ecosystems should not be used as an excuse to ignore the effects of fishing on those systems.

The effects of bottom trawling can be identified at a range of temporal and spatial scales. In assessing the significance of any given effect, consideration must be given to its relative scale (time and space), and the scale and distribution of the habitats and species which are affected. A species that is distributed over a wide geographical area will be less affected by an effect that operates only over a relatively small area. Similarly, broad-scale effects are more likely to adversely affect populations.

Effects have been grouped into two broad categories, those affecting the habitat itself (habitats), and those influencing species within habitats (species). In order to identify priorities for mitigation, we have ranked the effects within these categories. In several cases, we have been unable to rank the importance of some effects; in these cases we have given each the same rank. The criteria used to rank the effects are based on the consideration of:

- Temporal scale: Effects which cause permanent changes are of greater concern.
- Spatial scale. Effects which occur at large spatial scales are of greater concern.
- Direction of change: Decline in species or reduction in habitat features are of greater concern because of their potential irreversibility.

Other criteria could be considered, but WGECO felt that these three criteria were the most important with regard to evaluating biological impacts of fisheries. WGECO acknowledges the importance of social and economic effects, but were most competent to evaluate biological effects of fisheries on ecosystems. WGECO also notes that many international agreements (UNCED, FAO Codes of Conduct) consider that the pursuit of social and economic goals cannot be at the expense of conservation. WGECO also acknowledges the importance of the ecological consequences of any change and this aspect of ranking is addressed in later sections of the report.

6.3.2 Habitats

Bottom trawls can remove some physical features (Habitat Priority I)

Bottom trawls may cause the loss or dispersal of physical features in the environment such as peat banks or boulder reefs. These changes are always permanent, and lead to an overall reduction in habitat diversity. This in turn can lead to the local loss of species and species assemblages dependent upon such features, for example, attached bryozoan/hydroid turf and essential fish habitat. Even when substantial quantities of the habitat feature remain, if the habitat has become highly fragmented, this may compromise the viability of populations dependent upon it.

Bottom trawling can cause a reduction in structural biota (biogenic features) (Habitat Priority II)

Bottom trawling can cause the loss of structure-forming organisms such as colonial bryozoans, *Sabellaria*, hydroids, seapens, sponges, mussel beds, and oyster beds. These changes may be permanent, and can lead to an overall loss of habitat diversity. This in turn can lead to the local loss of species and species assemblages dependent upon such biogenic structures, for example, essential fish habitat for juvenile gadoids (Auster and Langton, 1999). The viability of populations dependent on biogenic features may be compromised even if the feature remains but has become highly fragmented.

Bottom trawling can cause a reduction in complexity (Habitat Priority III)

Bottom trawling can cause the redistribution and mixing of surface sediments as well as degradation of habitat and biogenic features. This can lead to a decrease in the physical patchiness of the sea floor (i.e., decreased heterogeneity) within fishing grounds. These changes are not likely to be permanent.

Bottom trawling can alter the detailed physical structure of the sea floor (Habitat Priority IV)

Bottom trawling can cause a reshaping of seabed features such as sand ripples, and damage to burrows and associated structures (e.g., mounds and casts, microhabitats). These features provide important habitats for smaller animals (meiofauna) and can be used by fish to reduce their energy requirements. These changes are not likely to be permanent.

An ecological consequence of all the changes in this section, which may result in habitat degradation, is reduced protection from predators of benthic species and the juveniles of commercially important fish and shellfish.

6.3.3 Species

Bottom trawling can cause the loss of species from part of their normal range (Species Priority I)

An extreme effect of bottom trawling impact can be the loss of species from part of their normal range for consecutive years or decades (extirpation). In many cases, it is not known whether species may recover their former range if fishing effects are removed. There may also be important consequences for genetic diversity of species, but in most cases genetic diversity has not been studied.

Bottom trawling can cause a decrease in populations which have low rates of turnover (Species Priority I)

The mortality inflicted on large-bodied species with low rates of potential population increase may lead to reduced population sizes for these species.

Bottom trawling can cause fragmentation of populations (Species Priority I)

Fragmentation of populations of species which have low mobility and limited dispersal of all life-history stages will compromise their ability to persist. Population fragmentation may have negative consequences for the genetic diversity of affected species.

The relative abundance of species is altered by bottom trawling (Species Priority II)

Community composition changes can be caused by bottom trawling. These changes can be increased or decreased relative abundance of species (diversity) depending on the nature of the unimpacted community, and how diversity is interpreted.

Fragile species are more affected by bottom trawling than robust species (Species Priority III)

Bottom trawls cause mortality. Some species are more likely to be killed than others, for instance, those with a fragile branching structure or a thin shell. Robust species may be better able to withstand this impact.

Surface-living species are more affected by bottom trawling than deep-burrowing species (Species Priority III)

Bottom trawls move over the sea floor and ground gear effects are confined to the top few centimetres of the sediment. Deep-burrowing species can avoid this effect.

Bottom trawling can have sub-lethal effects on individuals (Species Priority IV)

Sub-lethal effects of bottom trawling have been argued to include slow growth, increased likelihood of diseases, increased risk of predation, decreased competitive ability and, at a population level, reduction of average longevity.

Bottom trawling can cause an increase in populations which have high rates of turnover (Species Priority V)

Bottom trawling generally results in a reduction in predatory populations of larger fish and an increased availability of resources, such as food and habitat space. These may combine with the relative robustness of these taxa to trawl effects, and lead to larger populations although sometimes the effect is only local.

Bottom trawling favours populations of scavenging species (Species Priority V)

Bottom trawling activities have resulted in a greater availability of food items (e.g., injured and moribund benthos, discards and offal) for scavengers.

6.3.4 Food-web and ecosystem properties

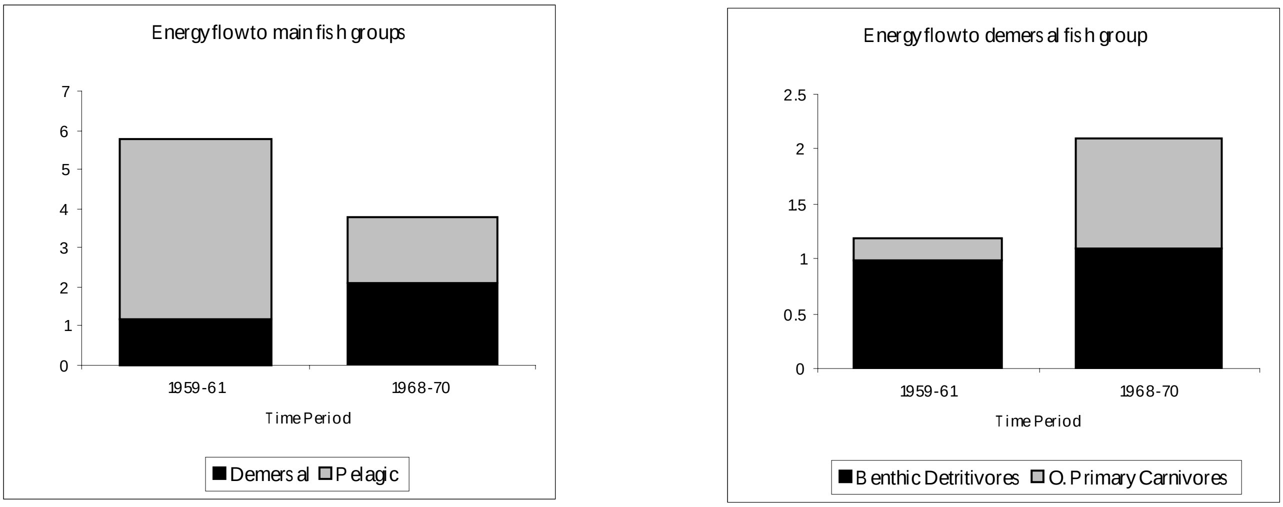
Fishing has had one major irrefutable effect on the food web structure of the North Sea: it has added a further highly efficient and effective ‘predator’ to the apex of the food web. Beyond this, the addition of fishing has served to alter the relative abundance of the species in the rest of the food web. The direct changes in species composition caused by gear-induced mortality and habitat loss have been discussed in depth in Section 6.2, above. These changes in species relative abundance could bring about changes in predator-prey interaction strength, and so alter the pathway of energy flow through the food web. Thus, for example, reductions in the abundance of large predatory fish species and carnivorous macrobenthic invertebrates could reduce predation on smaller benthic invertebrates, reducing energy flow from benthos to fish. In addition, increases in both the numbers and biomass of smaller, fast-growing benthic species could have the reverse effect of increasing energy flow rates to higher level predators. We contend that the alteration of food web structure is not in itself an immediate effect of fishing. Rather, it is a consequence of the changes in species composition that are brought about by fishing.

When considering the possible effects of fishing on food web structure, it is important to bear in mind that marine food webs are rarely stable. Environmental variation can result in considerable natural variation in the abundance of individual species. This is illustrated by the marked variation in 0-group fish production and the changes in seasonal immigration of *Calanus* into the northern North Sea. Such variation in species abundance is bound to alter the relative abundance of predators and prey, affecting individual predator-prey interaction strengths. Between-year, seasonal within-year, and spatial variations in the diets of fish and other top predators in the North Sea are readily apparent (e.g., Daan, 1989; Hislop *et al.*, 1991: Hislop, 1997; Greenstreet *et al.*, 1997), and in some cases such variation has been linked to variation in prey abundance (Tollit *et al.*, 1997; Greenstreet *et al.*, 1998). These changes in individual predator-prey interactions reflect natural variation in the food web energy flow pathway. It is against this natural background variation that any fishing-induced changes in food web structure must be judged.

Changes in food web energy flow pathways caused by fishing can be envisaged at two levels. Firstly, there are those where approximately similar quantities of energy continue to flow through groups of animals performing essentially the same ecosystem functional role. Only the identity of the species fulfilling these roles changes. For example, on Georges Bank, populations of gadoid species declined as a result of excessive fishing mortality during the late 1960s and early 1970s. This coincided with an increase in the biomass of dogfish, skates and rays, which at the time had no commercial value. Thus, the demersal fish assemblage, initially dominated by gadoid species, became increasingly dominated by these elasmobranch species (Fogerty and Murawski, 1998). In the North Sea, cod whiting and haddock stocks have also undergone considerable stock declines, leading to an estimated 60 % to 70 % reduction in the consumption of benthic prey by these predators. However, this reduction has been more than offset by increased levels of predation by flatfish, particularly common dab, whose population has increased markedly (Frid *et al.*, 1999). In both examples, demersal benthivorous or demersal piscivorous fish species have been replaced by other demersal benthivores or piscivores. Predator-prey interactions between individual species have changed, but the energy flow pathway between the major functional groups of the food web has remained relatively unchanged.

The second level of food web alteration involves changes where the energy flow through the food web takes entirely different pathways, involving groups of species performing quite different functional roles. One example has been documented for the North Sea where the food web, dominated by pelagic organisms up to the late 1960s, shifted rapidly to a demersal/benthic dominated system (Jones 1982, 1984). This shift had large consequences for the energy flow via particular functional groups through the system (Figure 6.3.2.1). Overall, the energy requirements of commercially exploited pelagic and demersal fish decreased by 35 %. The energy flow to the pelagic component of the system fell by 64 %, whereas flow to the demersal fish actually increased by 78 %. The source of energy for demersal fish also changed quite considerably. There was little change in the interaction between demersal fish and demersal detritivores, an increase of 11 %, but the rate of energy flow from the category 'Other Primary Carnivores' (e.g., euphausiids, sandeels, and 0-group gadoids) increased by over 400 %. Similar examples of food web structure changes can also be found for other regions, for example the Baltic Sea, where because of over-exploitation of cod, the food web changed from being dominated by cod to one dominated by sprat and herring (cf. Koster and Molleman, 1997). In this instance, the top natural predator in the food web was almost eliminated. A similar story has been suggested for the pikeperch-herring-zooplankton food chain. Modelling of this set of interactions suggests that current fishing mortality has reduced the pikeperch population to the point where marked cascade effects down the chain are likely to have occurred, resulting in increased herring biomass and decreased zooplankton biomass (Hansson *et al.*, 1997).

Figure 6.3.2.1. Change in the energy flow to demersal and pelagic fish between two time periods spanning the time of the herring fishery collapse.



It should be pointed out that the Baltic Sea is a species-poor system, with little or no redundancy of species within functional groups. Thus, any anthropogenic activity which causes major changes in the abundance of even a single species could have ramifications regarding the flow of energy between functional groups and so considerably alter the structure of the food web. Species-rich systems, with their functional groups comprising many species, should be more robust and resistant to such marked changes. Species replacement within groups should buffer against major changes in energy flow between groups. Yet the major shift from a pelagic-dominated to a demersal-fish-dominated North Sea food web coincided with the collapse of the North Sea herring fishery. Over-fishing undoubtedly contributed to this stock decline, but a sustained period of poor recruitment resulting from a major environmental change also played a large part (Corten, 1986, 1990). In this instance, the effect of severe over-fishing was perhaps to decrease the time frame during which the shift occurred. It could be postulated therefore that, had the environmental aberration been short-lived and the stock not badly over-fished, the shift may not have occurred. Continuing high fishing mortality among the adult herring caused the stock to collapse rapidly, allowing the regime shift to occur during the environmental conditions associated with poor herring recruitment. Had fishing mortality been lower, it is possible that with a return to more normal environmental conditions normal recruitment would have resumed, and would have stopped, or at least slowed, the population decline before the community changes became established. The conclusion to be drawn from such conjecture is that high fishing mortality may render food web structures more vulnerable to being disrupted in lasting ways by environmental variation.

A further point to make in this discussion is that in all the examples presented above, it was not fishing *per se* that caused the species replacements, or changes in functional group composition, resulting in changed food web structure. Rather it was *over*-fishing. Fishing mortality should be sustainable, thus reducing the likelihood of the type of stock collapse which is the precursor to major species replacement events. This is yet another reason why stock assessment, advice, and management all must take full account of recruitment variation and react swiftly to indications of poor recruitment. These should be intrinsic components of the precautionary approach.

Much of the above discussion revolves around what has been termed 'top-down' effects. It has been primarily concerned with the effects of changes in predator populations on the populations of their prey. However, bottom-up food web effects may also occur. Beam and otter trawling disturbs the seabed, causing nutrients locked up in the sediments to be released into the water column. This can stimulate local productivity, making more energy available at lower trophic levels. Community ecology theory suggests that such variation in resource availability may affect rates of energy transfer upwards, altering the relative abundance of predators in higher trophic levels. Such a response could cause an increase or a decrease in species diversity at higher trophic levels depending on local conditions. The consequences of such changes in the food web are uncertain. Related to this issue are the changes in benthic invertebrate species composition that have been described earlier in this report. Sections 6.2 and 6.3, above, suggest that species which are smaller bodied, grow faster, and have high reproductive capacity are likely to increase in abundance as a result of fishing disturbance, whereas species with the opposite characteristics are more likely to decline. This suggests that fishing may raise the productivity of benthic invertebrate components of the food web, potentially making more energy available to species higher up the food web.

6.3.4.1 References

Corten, A. 1986. On the causes of the recruitment failure of herring in the central and northern North Sea in the years 1972–1978. Journal du Conseil International pour l’Exploration de la Mer, 42: 281–294.

Corten, A. 1990. Long-term trends in pelagic fish stocks of the North Sea and adjacent waters and their possible connection to hydrographic changes. Netherlands Journal of Sea Research, 25: 227–235.

Daan, N. 1989. Database report of the stomach sampling project 1981. ICES Cooperative Research Report, 164. 144 pp.

Fogarty, M.J., and Murawski, S.A. 1998. Large-scale disturbance and the structure of marine systems: fishery impacts on Georges Bank. Ecological Applications, 8 (Supplement): 6–22.

Frid, C.L.J., Hansson, S., Ragnarsson, S.A., Rijnsdorp, A., and Steingrimsson, S.A. 1999. Changes in levels of predation on benthos as a result of exploitation induced changes in fish populations. Ambio, 28: 578–582.

Greenstreet, S.P.R., Bryant, A.D., Broekhuizen, N., Hall, S.J., and Heath, M.R. 1997. Seasonal variation in the consumption of food by fish in the North Sea and implications for food-web dynamics. ICES Journal of Marine Science, 54: 243–266.

Greenstreet, S.P.R., McMillan, J.A., and Armstrong, F. 1998.. Seasonal variation in the importance of pelagic fish in the diet of piscivorous fish in the Moray Firth, NE Scotland: a response to variation in prey abundance? ICES Journal of Marine Science. 55: 121–133.

Hansson, S., Arrhenius, F., and Nellbring, S. 1997. *In* Forage Fishes in Marine Ecosystems, pp. 281–291 Proceedings of the International Symposium on the Role of Forage Fish in Marine Ecosystems. Alaska Sea Grant Report 97-01, University of Alaska Fairbanks, Fairbanks, USA.

Hislop, J.R.G. 1997. Database report of the stomach sampling project 1991. ICES Cooperative Research Report, 219. 422 pp.

Hislop, J.R.G., Robb, A.P., Bell, M.A., and Armstrong, D.W. 1991. The diet and food consumption of whiting (*Merlangius merlangus*) in the North Sea. ICES Journal of Marine Science, 48: 139–156.

Jones, R. 1982. Species interactions in the North Sea. Canadian Special Publication of Fisheries and Aquatic Sciences, 59: 48–63.

Jones, R. 1984. Some observations on energy transfer through the North Sea and Georges Bank food-webs. Rapports et Procès-Verbaux des Réunions Conseil International pour l’Exploration de la Mer, 183: 204–217.

Koster, F.W., and Möllman, C. 1997. Predation by sprat and herring on early life stages of cod and sprat in the central Baltic: *In* Forage Fishes in Marine Ecosystems, pp. 41–69. Proceedings of the International Symposium on the Role of Forage Fish in Marine Ecosystems. Alaska Sea Grant Report 97-01, University of Alaska Fairbanks, Fairbanks, USA.

Tollit, D.J., Greenstreet, S.P.R., and Thompson, P.M. 1997. Prey selection by harbour seals *Phoca vitulina* in relation to variations in prey abundance. Canadian Journal of Zoology, 75: 1508–1518**.**

6.4 Mitigating the Effects of Bottom Trawls

The single most effective measure to protect marine ecosystems is to reduce fishing effort significantly and permanently. For some specific effects of fishing, some of the technical measures described here will be highly effective. However, for many effects of bottom trawls, most mitigation measures will only be effective at limited spatial scales, and only in combination with reductions in fishing effort. In many cases, failure to permanently reduce effort would mean that the benefits of mitigation measures, if implemented, would at best be temporary. Permanent reductions in fishing effort start to benefit populations quite quickly, whereas benefits for habitats, particularly those which are complex and composed of long-lived biota, will take longer.

There are a number of common sense issues which need to be considered in relation to any attempts at mitigating the effects of bottom trawling. These include:

- Recovery of populations and habitats from the effects of fishing may take from weeks to centuries. It requires reduction and, in some cases, cessation of the fishing activities which caused the effect for the full period of recovery, and perpetuating the recovery requires that this reduction or cessation is made permanent.
- There is a relationship between the intensity of trawling and the degree of change in the benthic environment.
- All technical measures are specific to a species or habitat, the scale at which they are employed, and their duration. No technical measure has generic use for all species and habitats.
- Simultaneous application of appropriate technical measures is most likely to achieve optimal mitigation results, as they may act synergistically.
- Mitigating impacts may be encouraged by adding economic incentives for their implementation.

6.4.1 Gear substitution

Replacing bottom trawls (beam and otter trawls) with static gears (gill nets, tangle nets, long lines, pots, traps, etc.) may reduce the degradation of sensitive habitats and, in some cases, mortality of non-target species. Static gears usually are more selective and generate a smaller proportion of unwanted catch, and generally cause limited impacts to habitat features. However, some of these gears may have unwanted effects on the ecosystem such as by-catch of seabirds and cetaceans, and therefore may be considered inappropriate in some cases. Reducing the impact of mobile gears in this way could provide full and effective protection for habitats (Table 6.4.4.1). The continued mortality caused by static gears to cetaceans, fish, and bird populations suggests that the benefits for this part of the marine environment may be more limited.

6.4.2 Gear usage

Altering the way in which bottom trawls are operated (e.g., requiring them to be towed for a shorter duration to aid survival of by-catch, or by on-board sorting and rapid return of by-catch to the sea) may, in principle, reduce the impact of the gears. It is recognised that, because of difficulties in enforcement, it is particularly important to obtain the agreement and understanding of fishers. Any gear modifications which encourage fishers to change their use of the gear to ways which dissipate the benefits, will ultimately not be of value. These measures are ineffective in reducing impacts on habitats, and are only marginally effective in reducing impacts on populations (Table 6.4.4.1).

Table 6.4.4.1. Summary of WGECO opinion on the relationship between effects of bottom trawling within existing environments in the Irish Sea and North Sea and possible management actions, within existing management frameworks. For mitigation measures which are not inherently spatial, the degree of mitigation is proportional to the extent of implementation. In assessing their effectiveness, we have judged each measure on its potential to provide protection against the effects listed and identified in Section 6.3. For mitigation measures with an inherent spatial dimension, we have assumed that 'Spatial Closure' will operate at the scale of four ICES rectangles. Other measures with a spatial dimension have been assessed assuming that they are implemented at an appropriate operational scale (i.e., real time closures triggered by unacceptably high by-catch would cover the area where the effect was occurring).

	Mitigation proportional to extent of implementation						**Mitigation with inherent spatial dimension**			
Effects (see Section 6.4)	**Reduce effort[1]**	**Gear substitution[2]**	**Gear usage[3]**	**Gear mod. (light/ novel)[4]**	**Gear mod. (select)[5]**	**By-catch quota[6]**	**Spatial closure**	**Real time closures**	**Improve habitat**	**Species adjustment**
Ai) physical*	-	C	-	-	-	-	C	-	C	-
Aii) biogenic	-	C	-	-	-	-	C	-	E	M
Aiii) complex	E	C	-	M	-	-	C	-	M	-
Aiv) structure	E	C	-	M	-	-	C	-	-	-
Bi) range	E	E	M	M	M	M	M	-	M	M
Bi) low turnover	E	E	M	M	M	M	M	-	M	M
Bi) fragment	M	E	M	M	M	M	-	-	M	-
Bii) relative	M/E	E	-	E	E	-	M/E	M	-	-
Biii) fragile	E	C	-	M/E	-	M	M/E	M	M	-
Biii) surface	E	C	M	M/E	-	-	M/E	M	-	-
Biv) sub-lethal	E	C	M	M/E	E	-	M/E	M	-	-
Bv) small Spp.	M/E	E	-	M/E	M	-	E	-	-	-
Bv) scavenger	E	C	M	M/E	E	E	M	M	-	-

- = none M = marginal protection E = effective protection C = complete protection *=no natural recovery potential

[1] Assuming a 50 % reduction in effort.

[2] Assuming full substitution of present demersal gears in enough areas that it results in reductions in sea floor impacts.

[3] Assuming changes are made in the way current gears are used in such a way as to reduce discard mortality.

[4] Assuming modification of gear to reduce impact on seafloor.

[5] Assuming modifications such as excluder/escape devices which increase selectivity, and the survival of species.

[6] Assuming quotas are set at an appropriate level to provide protection of vulnerable populations.

6.4.3 Gear modification

6.4.3.1 Lighten the gear

Increasing the weight of gear components (e.g., tickler chains) is used as a means to improve the efficiency of some gears which stimulate target organisms from the seabed. Greater weight at a given speed will increase habitat damage. Modifications to such gear to reduce their weight will have implications for the towing speed and the catch efficiency, and may have unpredictable consequences for species.

6.4.3.2 Use novel techniques

Modifications to bottom trawls can incorporate new technologies or designs to reduce the effect of gear components which contact the seabed. Any developments of novel gear should not be allowed to further impact the marine environment or spread such effects to previously inaccessible areas.

These mitigation measures are likely to be effective in offering marginal or effective protection for small species and scavenging species, but may be largely ineffective on many populations of fish (Table 6.4.4.1).

6.4.3.3 Develop more selective gears

Size selectivity of bottom trawls can be enhanced by a wide range of alterations to gear design. Most fisheries are multi-species and species selectivity can also be achieved by gear design. By these methods the mortality both of non-marketable sizes of target species and of non-target species can be reduced. Particular attention must be given to monitoring, and reducing as far as possible, post-selection mortality since regulation of gear design to improve selectivity assumes that escaping fish survive to benefit the fishery at a later date. Decreases in fishing mortality on non-commercial species will have positive effects on populations, but will be largely ineffective in reducing the impact of gears on habitats (Table 6.4.4.1).

6.4.4 Spatial closures

The complete closure of parts of the sea to exclude specific gears can protect areas and allow regeneration of impacted communities. Closures can be for a period of a few months or a full year. Full regeneration is only likely to be achieved if closures are permanent, and this will vary depending on the effectiveness of enforcement, and the ability of the communities to become re-established. Closures alone cannot repair many types of physical damage, but even on a small scale can provide full protection for uncommon and vulnerable habitat features (Table 6.4.4.1).

Closed areas can also reduce mortality of populations and increase productivity of species dependent on specific habitats, and in some cases of commercial species. The scale of the closed area will partly depend on the biology of the organism (e.g., rate and distance of migration). For fish there is evidence that the closure of nursery grounds where juveniles congregate may be more effective than closure of their spawning grounds (Fogarty and Murawski, 1998).

Spatial closures which result in the relocation of fishing effort to other areas may cause additional problems in those areas. Therefore, spatial closures are most effective when accompanied by effort reductions. The balance of benefits from spatial closures needs to be evaluated on a case-by-case basis, and difficulties may be partly resolved by area zonations.

The effectiveness of closed areas will be greatest on habitats and for species which are relatively immobile. The effectiveness of the closure will also relate to the spatial scale of the closure in relation to the spatial extent of the feature or population. Clearly, those areas which are permanently closed for a full 12 months (i.e., not seasonal), are likely to be most effective. Despite these obvious difficulties, for the purpose of protecting fragile habitats, closed areas are likely to be one of the most effective measures, especially when combined with reduced fishing effort.

6.4.5 Real time closures

The temporary closure of an area at which high by-catch rates occur (real time closure) can be an effective tool in fisheries management. These measures have marginal benefits for some species in the benthic community, but will generally be site-specific and short-term (Table 6.4.4.1). They will not have clear benefits for habitats. The effectiveness of these closures may be limited to commercial fish stocks, and require considerable investment in monitoring and enforcement to realise benefits.

6.4.6 Improve habitat

Degraded habitats can be improved by artificially restoring features of the physical environment, and this could help to reduce the effect of bottom trawls (Table 6.4.4.1). This will only be a short-term measure unless the cause of the impact is permanently removed, and costs may prohibit such action on a large scale.

6.4.7 Species adjustment

Considerable time and expense have been invested in studies on the potential for artificially enhancing depleted stocks of commercial fish and shellfish. Despite the effort involved, the success of these cultivation and release programmes has been difficult to measure and results are equivocal. Even if these programmes can be shown to successfully augment natural populations, they will be a short-term measure unless the cause of the impact is permanently removed, and the direct mortality reduced to a sustainable level. Little effort has been expended on the artificial enhancement of populations of other benthic biota.

It has been suggested that the imbalance to marine ecosystems caused by human exploitation can be artificially re-adjusted by culling or harvesting abundant species. Attempts at culls of abundant populations in terrestrial ecosystems, whose dynamics were better understood than those of most marine systems, often had different consequences from those desired, and did not provide the expected benefits. These results argue strongly against considering culls as a general tool for ecosystem management.

Any gain from enhancing or culling species is likely to be short-term and be of very limited effect (Table 6.4.4.1).

6.4.8 By-catch quota

This is similar to a catch quota in that it sets a limit on non-targeted marine organisms (either fish or other benthic species). Reaching the quota triggers the closure of the fishery. Mechanisms to set and enforce such quotas have not yet been developed in the EU but have been used off Alaska and elsewhere. Managing with by-catch quotas usually means closing the fishery before the quota of the directed species has been taken, however, there are practical difficulties with implementation and enforcement. Gains from by-catch quotas might be greater for species at particular risk from trawling, and may well reduce food availability for scavengers (Table 6.4.4.1).

6.4.9 References

Auster, P.J., and Langton, R.W. 1999. The effects of fishing on fish habitat. *In*: Fish habitat: essential fish habitat and rehabilitation, pp. 150–189. Ed. by L. Benaka. American Fisheries Society, Symposium 22, Bethesda, Maryland, USA.

Daan, N., and Richardson, K. (Eds.) 1996. Changes in the North Sea ecosystem and their causes: Århus 1995 revisited. ICES Journal of Marine Science, 53: 1–1225.

Fogarty, M.J., and Murawski, S.A. 1998. Large-scale disturbance and the structure of marine systems: fishery impacts on Georges Bank. Ecological Applications, 8 (Supplement): 6–22.

6.5 Achieving Reductions in Effects

6.5.1 Context

WGECO calls attention to two types of changes which are required in order to achieve meaningful reductions in the effects of bottom trawls on the benthos of the North Sea and Irish Sea (and more generally, to contribute to the reduction of ecosystem effects of fishing). We advise a series of measures below, which, if implemented successfully, would reduce the effects which we have considered to be the most serious. The advised measures have been developed for the existing framework of governance, including but not restricted to, the Common Fishing Policy of the EU. We also advise on changes to the governance framework itself, which we consider would be necessary to successfully implement most of the advised measures, and enhance conservation of North Sea and Irish Sea marine ecosystems.

None of the priority management measures are fully developed, with specific areas, species, fleets or magnitudes of change specified. To address area issues with specificity would require detailed maps and data not available to WGECO. More importantly, however, to prepare recommendations to enhance the conservation of specific species, habitats, or sites, or to change harvesting opportunities among fleets would require clear priorities among different species, habitats, fleets, etc. Biological considerations have a role in choosing these priorities, but the task can only be done in conjunction with many other parts of society (see Section 7, below). However, until such priorities are set, specific advice can only be provided in response to specific questions. Given a specific question and necessary data on the specified habitat, area, or species, the advice below should provide the framework for detailed advice on management actions.

6.5.2 Priority management measures

The order of presentation reflects the priority given to each measure by WGECO. We stress that the value of any measure depends on the quality of its design and implementation.

6.5.2.1 Major reduction in fishing effort

Almost all the effects of fisheries on species would be reduced if fishing effort overall, and specifically with bottom gears, were reduced substantially. There is no specific percentage reduction which can be advised globally, but generally WGECO considers reductions of more than 30 % to be the minimum likely to have any measurable benefits, and often even greater reductions would be required. Of the effects WGECO considered most serious, major reductions in effort would contribute greatly to reducing the effect of reducing the range of species, and be of some help in reducing effects on species with low turnover rates and species suffering fragmentation of range. Such effort reductions in bottom trawl fleets would also reduce many other species effects of concern, including effects on relative abundance of species, on fragile species, on surface-living species, and reduce sublethal effects. Many but not all of those groups would benefit from comparable reductions in other types of gears as well.

Effects of bottom trawls on habitats would not be greatly reduced by reductions in bottom trawling alone. However, effort reductions would often be necessary to realize full benefits of other measures proposed below to reduce effects on habitats.

At current fishing mortalities, effort could be reduced greatly without the fisheries suffering major reductions in catches. Moreover, the direct benefits to the target stocks would often result in increased catches in future years. WGECO stresses, however, that effort reduction must be effective. Simply limiting a measure of effort such as hours fished is unlikely to result in genuine reductions in fishing pressure, due to the likelihood of changes in fishing behaviour to maintain catches in response to the regulation.

6.5.2.2 Closed areas

Permanent year-round closed areas can fully protect specific valuable and vulnerable habitat features, if the closed area is sited correctly and closed effectively. The size of the closed area will depend on the scale of the feature which is to benefit from the closure. There may be least opposition to closing areas which have received little fishing pressure historically, and may have suffered the least disturbance, but priority should be given to habitats of highest value and vulnerability. Closures of areas used by fisheries could displace that effort into other areas. Programme to ensure that the displaced effort is either removed from the fishery, or relocated to areas where new damage will not result, must be part of the implementation of closed areas. Effective closed areas eliminate effects of trawling on physical and biogenic features within the closed area, and reduce or eliminate effects on habitat complexity as well. Sedentary species within closed areas also benefit from the closure, so this measure can reduce effects on low turnover species, fragile species, surface-living species, and sublethal effects of sedentary species within the closed area. Population fragmentation and range reductions may or may not benefit from closed areas, depending on the mobility of the species and the location of the closed area relative to remaining populations. As explained in past advice (ICES, 1994), permanently closed areas would also have many benefits for research to improve our understanding of natural ecosystem dynamics and, hence, for our ability to evaluate the effects of fishing on ecosystems.

Short of full closure, zoning (in the EU meaning of partitioning an area and segregating gears into different portions) can functionally exclude bottom gears from parts of their range of operation. In this way, their benefits are very similar to closed areas for habitats and sedentary species, unless the zoning system allows operation of other gears with ecosystem impacts similar to bottom trawls.

Closed areas do not reduce the overall opportunity to harvest commercially important fish, but they often concentrate fishing relative to prior to closure. Consequently, associated measures intended to prevent excessive concentration of effort outside the closed area or relocation of effort to new areas may in practice result in reduced opportunity to fish.

6.5.2.3 Gear substitution

Gear substitution may or may not have major benefits in reducing the effects of bottom trawls on species suffering range reduction, and species with low turnover rates, depending on the vulnerability of the species of concern to the new gear. In cases where by-catch of these species is low in the alternate gear, benefits can be large. Relative abundance of species will still be changed by fishing with the substituted gear, but in different ways depending on the differences in selectivity of the new gear compared to the bottom trawl. Reductions in effects on habitat features may be achieved by gear substitution, but the substitution must be sufficiently widespread that it results in noteworthy reductions to the portion of the sea floor that is impacted by mobile bottom gears. Under those circumstances, there can be major reductions or elimination in effects on fragile and bottom-living species as well.

Gear substitution does not reduce the overall opportunity to catch fish, but usually redistributes those opportunities in ways which can have large economic or social impacts.

6.5.2.4 Gear modification

There are cases in which gear modifications have been very effective in reducing direct mortality on specific by-catch species, for example, turtle exclusion devices. Moreover, gear modifications which reduce the contact of bottom trawls with the benthos can reduce most of the species effects, but the reductions will be case specific. The potential depends greatly on the behaviours of both the benthic species of concern and the target species; it has to be possible to maintain a high catchability of the target species while reducing mortality inflicted on the species of concern. Unless impact of the gear with the bottom is largely eliminated, effects on habitats would continue.

Gear modification does not have to reduce opportunities to catch fish, but may result, or be perceived as resulting, in lower efficiency, and could prompt harvesters to compensate by increasing effort. Actual effects will be highly case specific.

6.5.2.5 Habitat rehabilitation

This measure could address specific effects of bottom gears on physical features of habitats. Benefits would be local and require continuous protection from further damage by fishing gears, if the benefits were to persist after the rehabilitation was implemented. It is advised for very specific needs, such as possibly replacing the boulder fields of the southern North Sea, or reintroducing oyster beds in areas where they have been lost.

Habitat rehabilitation will not reduce opportunities to catch fish, but usually will be associated with prohibitions of at least trawling within the rehabilitated areas.

6.5.3 Governance changes

We acknowledge that these are not novel proposals, although the justification for each, contained in earlier parts of Section 6, may be more complete than before. Because they are not novel proposals, we asked why they have not been implemented already. We identified clear institutional and governance impediments to implementing the types of measures advised above. These also must be confronted, if the effects of fishing on ecosystems are to be reduced. The review of the Common Fishery Policy in 2002 presents an excellent opportunity to confront these issues.

It is essential that competent management agencies set explicit operational objectives for conservation of habitats and target and non-target species affected by fishing. This will require wide consultation, but without clear and ordered objectives it will never be possible to rank conservation actions with consequences for different parts of the ecosystem. The effectiveness of many of the possible management measures for dealing with the effects of fishing would increase greatly if there was effective on-board monitoring of fishing operations. This programme would also benefit achieving existing fisheries management objectives, such as quota compliance.

In addition to dialogue and action with a wider range of professionals and stakeholders with regard to operational objectives and priorities, additional interaction with these groups is essential on two other fronts. One is to develop economic incentives to facilitate implementation of and compliance with management measures intended to reduce the effects of fishing on ecosystems. The other is to explore ways to increase the credibility, and often breadth and reliability, of scientific advice on these complex issues. This should include consideration of making more extensive use of quantitative and qualitative information from resource users, and incorporating them, where appropriate, in evaluating the status of species, habitats, and ecosystems.

6.5.4 Specific immediate actions

Each of the recommended courses of action listed above clearly requires significant effort to bring existing knowledge to bear on specific problems, and in many – possibly all – cases, to acquire substantial new knowledge. This will take time. Nonetheless, at this time WGECO feels that there are particular habitats and species currently not being conserved. The Precautionary Approach requires that immediate action be taken in the cases below, to ensure that conservation is not compromised while greater knowledge bases are being built. As management plans are being developed for habitats in urgent need of protection, and recovery plans are being developed for species in urgent need of conservation action, progress can be made on improving our understanding of the threats to the habitats or species, and refine the measures used to ensure their conservation.

6.5.4.1 Prevent expansion of areas impacted by bottom trawls

Restrict the use of each bottom trawl gear presently in use in the North Sea and Irish Sea to those areas where it is presently employed. This will protect habitats and prevent fishing mortality on sedentary benthic species in areas not yet exposed to gears currently in use or recovering from effects of historic trawling. This would be a step towards zonation of the North Sea and Irish Sea, particularly if area expansion of other types of gears was also restricted.

6.5.4.2 Prevent expansion of the numbers of bottom trawlers

Restrict the numbers of licenses for the various types of trawl gears to no more than the current fleet sizes. Allow transfers of licenses only to gears which are documented to have lower impacts on habitats, or no great impacts on habitats and lower by-catch. This will prevent the effects on the ecosystem from increasing, and ensure that the direction of change in fleet characteristics is toward fleets with lower effects on ecosystems. Similar measures would be appropriate for other types of gears as well, to help achieve the necessary reductions in effort and reduce by-catches of demersal and pelagic species. However, such restrictions would be less likely to reduce effects on benthos, and are technically outside the scope of the request for advice.

6.5.4.3 Strengthen interactions with groups working on conservation of these ecosystems

Competent management agencies should cooperate fully with the spectrum of initiatives now under way to identify and protect species and habitats at risk in the North and Irish Sea (and elsewhere). As these initiatives identify specific species or habitats that require conservation action, management agencies should implement effective conservation measures swiftly. The types of initiatives of particular interest to WGECO include, but are not restricted to, components of Natura 2000, including biodiversity action plans, national biodiversity inventories, and other programmes to implement the Jakarta Convention. **Management agencies should begin now to develop the legal framework and enforcement mechanisms required to act quickly as specific conservation measures are required.**

6.5.4.4 Improve ability to detect and measure impacts

Increased support should be made available to develop instrumentation and monitoring programmes to measure the effects of fishing on benthic ecosystems, and the consequences of measures intended to reduce the effects. Better quantification of effects is needed to enable conservation requirements to be identified, to allow benefits of actions to be measured, and to provide more convincing evidence that specific conservation measures are warranted.

6.5.5 Reference

ICES. 1994. Joint Report of the ICES Advisory Committee on Fishery Management and the Advisory Committee on Marine Environment, 1994. ICES Cooperative Research Report, 203. 21 pp.

7 ECOSYSTEM MANAGEMENT OBJECTIVES

7.1 Introduction

WGECO began a consideration of the issue of ‘reference points which include ecosystem considerations’ in 1997. We would begin by reiterating our assertion that management of fishing effort at levels which deliver a high probability that conservation objectives are achieved for the target stocks (i.e., $SSB > B_{pa}$) is likely to be the single biggest change that would ensure conservation of the ecosystem. This is especially so if combined with targeted protection of key habitats/features.

OSPAR and the North Sea Conference of Ministers consider the implementation of an ecosystem approach in fisheries management as an important step for the integration of fisheries and environmental issues. Since there is not yet a clear and agreed upon definition of an ecoystem approach (NRC, 1999), the approach taken in this chapter will be along several lines. There are a growing number of documents which describe the features of ecosystem management approaches (e.g., Anon. 1995; Christensen *et al.*, 1996; Lanters, 1999).

In reviewing this issue in 1997, WGECO identified the following areas where additional reference points might be appropriate:

- reference points for non-target species;
- reference points for ecologically dependent species (species that are so tightly linked ecologically to the target species that changes in the abundance/distribution of the target, which do not approach $B_{pa,}$ may still compromise the status of the ecologically dependent species—ICES, 1998a, Section 8.2.2.2);
- reference points for the genetic health of populations;
- reference points regarding the consequences of increased populations of scavengers.

In this section we revisit these considerations in the light of advances in our understanding which have emerged since 1997 and add a consideration of other areas of the ecosystem where management may be required to achieve sustainability and where reference points may be defined. Specifically we will consider the need and possibilities for reference points for single species including, but not restricted to, scavengers, marine habitats, genetic health of populations, and ecosystem properties.

Such considerations implicitly recognise the need for integrated management of the marine environment and that managers will have to operate with multiple management criteria. Such multiple management criteria are now an accepted part of fisheries management in multispecies fisheries such as in the North Sea, even if methods for simultaneously meeting them all are not perfected.

In developing an ecosystem scale management perspective it must be recognised that the objectives set will include much wider considerations than those traditionally addressed for fisheries management. The overall ecosystem objective should involve sustainability. Sustainability means different things to different people. We take it to mean that current activities do not compromise the ability of the environment to provide resources and services in the future, nor reduce the choices available to future generations. Further, we should recognize that with regard to fisheries there are three aspects to sustainability:

1) Sustainable fisheries. The level, and composition, of landings are sustainable.
2) Sustainable fishing industry. This is the socio-economic sustainability of fishing and includes considerations of the viability of communities dependent on fisheries, the size and nature of the fishing industry and all linked economic and social activities—including merchants and fish processing sectors, chandlers, vessel building and repair, etc.).
3) Sustainable ecosystems. The nature, species composition and functioning of the environment are not placed at risk of changes that seem long lasting and difficult to reverse.

It is not for scientists to advise on the balance between these three, but such a consideration must form an explicit part of any ecosystem management scheme. It should however be recognised that a number of existing international agreements (Table 7.1.1) already place a priority on sustaining the ecosystem, arguing that pursuit of social and economic sustainability cannot be allowed to result in an unacceptable risk to conservation of the ecosystem.

Any ecosystem approach to management must also have mechanisms for dealing with the inherent uncertainty in predictions of marine system dynamics. The application of a precautionary approach to fisheries management has seen advances in recent years but these will need to be developed and extended if any management scheme based on an ecosystem approach is to be effective. In particular, admonitions that uncertainty about the status of single species cannot be used as a reason to defer cost-effective measures to reduce risk, must be expanded to acknowledge the greater uncertainty about ecosystem status and trajectory.

WGECO stresses that science has to deal with the complexity of the marine system that includes thousands of species and many different types of habitats. The degree of mutual coherence is poorly known and predictive scientific models are not, and may never be, available. In addition, human use may already have changed the most sensitive components of the marine system, hampering identification of reference levels. If any changes are observed in ecosystems it is important to differentiate between changes that form part of natural variability and those that represent the effect of one or more human activities. In is in this context that operational reference points are considered for species, habitats, genetics, and emergent properties of ecosystems.

Table 7.1.1. An overview of the main global conventions, laws and treaties applying to the conservation and management of marine living resources. These are often given regional specificity in ‘local’ conventions such as Annex V of the OSPAR Convention which covers protection of species and habitats.

Convention or treaty	Year	Main objective
UN Law of the Sea	1982	Regulation of the management and authority of all living marine resources. Establishment of an Exclusive Economic Zone
Bonn Convention	1983	Protection of migratory stocks of wild species (species moving across national borders)
CITES and GATT		General Treaties governing prevention of trade in endangered species (CITES) on reduction of environmental impact (GATT)
Convention on Biological Diversity (CBD)	1992	Result of UNCED Conference. Protection of biodiversity at level of genetics, species and ecosystems
Agenda 21 - Chapter 17	1992	Result of UNCED Conference. Protection of all marine and coastal areas by rational use and development of living resources
FAO Code of Conduct	1995	Code of Conduct for Responsible Fisheries by considering ecosystem and socio-economic aspects of fisheries and the precautionary approach
Jakarta Mandate	1997	Elaboration of CBD for marine systems in which Marine Protected Areas form a major issue
UN Convention on Migratory and Straddling Fish Stocks	not in force	Conservation and protection of border crossing and high seas fish stocks

7.2 Population and Species Reference Points/Objectives

What is at risk and how do fisheries place them at risk?

7.2.1 Populations of target and non-target species

If improperly managed, fisheries can place **populations of both target and non-target species** at risk, through inflicting unsustainable mortality over periods of time long enough to impact abundance. The mortality can be severe enough to cause a population decline directly, to spawning biomasses at which either the probability of good recruitment is reduced or the probability of poor recruitment is increased. These are the criteria presently used by ICES to decide if a stock is inside or outside safe biological limits.

Where fisheries inflict less severe mortality, the fishery will change **the age composition of the stock** relative to the unexploited condition. The changes may be great enough that spawning biomass comprises disproportionately first-time spawners or total biomass may depend excessively on new recruits. Neither of these changes is desirable, as there is evidence that for at least some species first-time spawners have lower reproductive value on a per kilogram basis (Trippel, 1998), and dependence of biomass in incoming recruitment makes the stock more vulnerable to short-term periods of poor recruitment or environmental stress. Hence, reference points even for target species should ensure a suitable age composition as well as adequate total spawning biomass and sustainable fishing mortality.

Without being killed, target or non-target species may suffer **injury or exposure which results in increased vulnerability to predation**. This can result from physical damage as gear passes over individuals or as individuals pass through gear, or from rough handling and release. Once injured or exposed, if predators are present, the biological effect is much the same as for direct mortality. Hence, the seriousness of this effect would be evaluated in the same way as for direct fishing mortality: is the total death rate sustainable and is biomass being conserved?

NOTE: For all the direct effects above, from a biological perspective, and according to the international agreements reviewed in Table 7.1.1, ICES is concerned about the conservation of all species. Hence we ask the same questions about the sustainability of all populations in the face of total mortality and the contribution of fishing mortality to total mortality. **There is no justification to apply different standards to species of commercial and non-commercial importance.**

The direct mortality due to being killed by fishing gear can become excessive if effort is too high, either overall, or in the area where the species suffering unsustainable mortality is concentrated. Injury or exposure by gear that results in increased vulnerability to predators can jeopardise conservation of a species if a biologically important fraction of a population encounters the gear and is not retained.

7.2.2 Spatial properties

Fishing can successively deplete meta-populations so that even if local subpopulations are not demonstrated to be genetically distinct, the species or stock ceases to be present in progressively larger parts of its historic range. Although special circumstances would be required, it is theoretically possible that a population as a whole could be above its biomass reference point and experiencing total mortality still below the mortality reference point, yet the fishery could be causing a reduction in range. The key circumstances include intense localized exploitation and low mobility of the species being killed. There are several reasons that managers should take safeguards that fisheries do not cause major reductions in range. It has been theorized that a species (or stock) becomes less resilient to environmental challenges as distribution contracts, if only by becoming more vulnerable to catastrophies (Tuljapurkar, 1990). Some studies have conjectured that a reduction in spawning area reduces reproductive potential by not allowing full seeding of larval/juvenile habitats (Burgman *et al*., 1992; Groom and Pascual, 1998). Also as a population becomes spatially concentrated, q (catchability to fishing gear) goes up and the stock becomes more vulnerable to further overfishing, even when fleet behaviour has not changed.

Reduction in range or in meta-population structure can occur if a fishery is not distributed representatively across the full range of the species of concern, and redistribution of the species is slow relative to its population dynamics responses to fishing mortality.

7.2.3 Dependent species

Fishing can deplete a population locally so dependent predators cannot find sufficient food to survive or reproduce at sustainable rates, even though the stock as a whole is within safe biological limits, and the population genetic diversity may not be compromised. Evidence for this effect, and reasons to be concerned about it, are reviewed in ICES (1999b).

Conservation of ecologically dependent species can be jeopardized if the fishing fleet is more mobile than the dependent species and the prey is widely distributed but slowly mobile. Given those two factors, a fishery may cause local depletions of prey for periods of time that are long relative to the needs of the dependent species, if the fishery concentrates harvests disproportionately in areas important to the dependent predator.

7.2.4 Scavenger-caused effects

Fishing can produce so much waste that species which feed on offal and discards can increase greatly in abundance. The incidental mortality that the scavenging species inflict on alternate prey may become unsustainable, or through competition for limited space the scavenging species may cause reproduction below replacement rates for the species displaced from breeding (or other) sites. Evidence for this effect is reviewed in Section 6.2.

Fishing produces wastes (discards and offal) which can be concentrated and readily available as food for scavengers who can exploit this food source. If the scavengers also prey on species that cannot use this food supply, or compete with them for breeding space, then a fishery that increases food to scavengers may cause mortality or poor recruitment of species who are eaten or out-competed by scavengers.

7.3 Habitat Features

What is at risk and how do fisheries place them at risk?

Marine habitats are generally distinguished by the physical nature of the environment; e.g., silty-mud is distinct from muddy-sand, frontal regions separating mixed and stratified waters. These can include biologically produced features such as reefs and turf.

Changes in the nature, extent and spatial distribution (degree of patchiness) of habitat features can compromise the ability of the ecosystem to support a natural species assemblage and hence normal ecosystem function (Dayton *et al.*, 1995; NRC, 1999).

There are limited data on the impact of fishing on habitats within EU waters (see Section 6.2). In addition to the impacts recognised from bottom trawls (Section 6.3), there are data which suggest that deep-water fisheries to the west of Scotland, around the Faroe Islands and in northern Norway have caused substantial damage to beds of the cold-water coral *Lophelia*, and data also indicate damage to *Sabellaria* reefs in coastal waters of the North Sea and Irish Sea (STECF, 1999).

In the northwest Mediterranean, changes in the size and species composition of fish populations caused by fisheries may have led to large changes in benthic communities as a result of increased abundance of sea urchins (Sala *et al*., 1998). This is an example of a habitat modification mediated through changes in the food web.

Section 6.3.1 presents the conclusions about aspects of marine habitats which may be put at risk by fishing. They are repeated briefly here.

Bottom towed gears can remove some physical features

Bottom towed gears may cause the loss of physical features in the environment such as peat banks, boulder reefs, or gravel banks. These changes are always permanent, and lead to an overall reduction in habitat diversity. This in turn can lead to the local loss of species and species assemblages dependent upon such features. Examples might include attached bryozoan/hydroid turf and essential fish habitat such as herring spawning grounds. Even when substantial quantities of the habitat feature remain, if the habitat has become highly fragmented, this may compromise the viability of populations dependent upon it.

Bottom towing of gears can cause a reduction in structural biota (biogenic features)

Loss of structure-forming organisms such as colonial bryozoans, *Sabellaria*, hydroids, sea-pens, sponges, mussel beds, and oyster beds can result from the impact of bottom towed gears. These changes maybe permanent, and lead to an overall loss of habitat diversity. This in turn can lead to the local loss of species and species assemblages dependent upon such biogenic structures. Essential fish habitat such as juvenile gadoid nursery habitat would be an example. Even when substantial quantities of the biogenic feature remain, if the feature has become highly fragmented, this may compromise the viability of populations or species dependent upon it.

Bottom towed gears can cause a reduction in complexity

Towing of bottom fished gears can cause the redistribution and mixing of surface sediments as well as degradation of habitat and biogenic features. This can lead to a decrease in the physical patchiness of the sea floor (i.e, decreased heterogeneity) within fishing grounds. These changes are not likely to be permanent.

Bottom towed gears alter the physical structure of the sea floor

Towing of gears on the sea floor can cause a reshaping of seabed features such as sand ripples and damage to burrows and associated structures (e.g., mounds and casts, microhabitats). These features provide important habitats for smaller animals such as meiofauna.

7.4 Genetic Properties of Populations

What is at risk and how do fisheries place them at risk?

Total genetic variation within a species can be partitioned into variation within and among populations. Fisheries may have consequences for both types. Within populations, phenotypic changes associated with fisheries are well documented for a number of species and include changes in morphological and life history traits such as weight- and length-at-age, and age- and length-at-maturity, spawning time, etc., (e.g., Rijnsdorp, 1993; Rowell, 1993; Millner and Whiting, 1996; Trippel *et al.*, 1997), many of which may be correlated (ICES, 1997). Such changes may arise through relaxation of intra-specific competition, response to shifts in environmental conditions (phenotypic plasticity) and to change in genetic composition; it is often difficult to establish which of these effects is responsible for the observed response. To the extent that the changes are genetically based, intensive selective fishing will result in changes in gene frequencies, and possibly in loss of alleles within the exploited populations.

Populations that are reproductively isolated, with little or no gene flow between them, will tend to diverge genetically either through different selective forces or through genetic drift. Salmonids have high among-population variance resulting from their homing behaviour at spawning time (e.g., Gharrett and Smoker, 1993). However, even in species that have free-drifting larvae, gametes or spores (approximately 70 % of marine invertebrate species have pelagic larvae; Mileikovsky, 1971) and are ultimately distributed over a wide area, local populations can often be discerned (e.g., cod: Ruzzante *et al.*, 1997; squid: Shaw *et al.*, 1999; marine algae: Van Oppen *et al.*, 1996). In such species, loss of sub-populations results in loss of the unique characteristics of the genome of the sub-population.

Natural selection acts within populations, while the genetic potential of the species to adapt to environmental changes depends on the total genetic diversity represented among populations. It is necessary to maximize both types of variation to maintain full potential for evolutionary change within a species.

In general, modelling studies have shown that size selection favours slow-growing and late-maturing fish, although there are exceptions to this (ICES, 1997).

Fishing mortality is a highly selective process, both with respect to the size of the organism captured and location (ICES CM1997/F:4). The fishery may also directly or indirectly favour capture of one sex over another (e.g., American lobster, shrimp), altering the sex ratio and/or sex-specific size frequency of the breeding population. In addition, migratory stocks may be under different selection pressures in different parts of their range due to different fishing methods. Fishing therefore has the potential to affect the genetic diversity and genetic structure of a species.

Selective breeding programmes for cultured fish (e.g., salmon) and invertebrates (e.g., abalone) have shown that significant amounts of genetic heritability (the proportion of phenotypic variation that is inherited from one generation to the next) exist for yield-related traits. Life-history traits, being closely linked to fitness, have relatively lower heritabilities, however even these are capable of showing a substantial selection response in only a few generations (ICES, 1997). Although extrapolation of heritability estimates determined from breeding programmes to those in wild fish stocks should not be made, this research has demonstrated clearly that there is genetic variation in those traits selected for by fishing. The stronger the selectivity (in the fishery sense ‘selective’) of the fishery for certain traits, and the greater the proportion of total mortality made up of fishing mortality, the greater will be the effect of fishing on the genetics of the exploited population. The persistence of fishing-induced genetic changes will depend upon the other selective forces operating on the species, the proportion of genetic diversity affected and the reproductive biology of the species. In some cases, genetic change may not be readily reversed by altering fishing practices (Law and Grey, 1989). Consequently, fishing can cause evolution of phenotypic traits of the exploited species (Law and Rowell, 1993), although the time scale over which it operates is unknown.

Fishing can also selectively harvest some sub-populations intensively, while harvesting other sub-populations lightly. In these cases, a rate of fishing mortality which is sustainable at the scale of the whole species may successively eliminate isolated sub-populations, and reduce the total genetic variability of the stock or species.

7.5 Emergent Properties of Ecosystems

7.5.1 Emergent properties: What are they?

In previous reports we have considered ecosystem level reference points (ICES, 1996, 1998a). Discussions within WGECO highlighted issues such as:

- food web dynamics;
- species richness and evenness (diversity);
- distribution of life histories;
- production:biomass ratios.

These are not direct biological properties but are functions of the entire ecosystem and are referred to as emergent properties. They are important not only because they may tell us something about the functioning/status of the ecosystem but also as they have been widely perceived as indicators of environmental status.

7.5.1.1 Does fishing put emergent properties at risk?

There has been considerable speculation as to the extent to which fishing may alter these emergent ecosystem properties (see ICES (1998a) and earlier sections of this report). It is also true that many press and popular articles have been highly emotive in their commentary on this issue. We have reviewed the **evidence** that has emerged since our last consideration and can find none which would cause us to revise our conclusions.

WGECO stresses that the need for some ecosystem reference points is real. At this time WGECO believes that we are not in a position to recommend that ecosystem emergent property reference points are necessary, beyond the reference points which would assure sustainability and conservation of all species and habitats impacted by fishing. Neither are we prepared to confirm that single species, habitat and genetic reference points alone are enough to ensure a precautionary approach to ecosystem management. Some study may yet provide compelling evidence that reference points for emergent properties of ecosystems are also required to ensure conservation of the ecosystem, but to this time none have.

7.6 Reference Points for Management

Ecosystem approaches to marine management will require many reference points. Exceeding any reference point whether for target species, non-target species, habitat change or genetic health, should invoke mitigation measures.

7.6.1 Populations and species

7.6.1.1 Direct mortality

For target species conservation can be achieved by following the precautionary approach. Special importance should be given to two activities. One is setting B_{pa} and F_{pa} sufficently far from the biological limits to allow for uncertainty in estimates of present biomasses and fishing mortalities, and uncertainty about the future states of nature (especially, but not exclusively, future recruitment) for the time scale of management and the degree of risk aversion managers (and society) demand. The other is implementing harvest control rules, to ensure that necessary conservation measures are implemented in a timely way when a reference point is violated. Together, these measures should keep target species inside safe biological limits with high probability (ICES, 1998b). Occasionally the biology of a species makes an escapement goal or a total mortality a more appropriate reference point than an exploitation rate, but those circumstances are well understood (ICES, 1999b).

For non-target species there is no reason to take a different approach to assuring conservation. The implementation problem is the practical impossibility of setting biomass and fishing mortality reference points for every non-target species in the ecosystem, and then assessing compliance. As a practical solution we propose setting biomass and fishing (or total) mortality reference points for non-target species of high vulnerability, and monitoring their compliance. This proposal assumes that the documented conservation of a set of non-target species of high vulnerability gives high probability of also ensuring conservation of other non-target species of lower vulnerability. We suggest that vulnerability should be evaluated with regard to:

- the ability of the species to tolerate an increase in mortality (see Section 6.3—long-lived species of low fecundity are likely to be more vulnerable than short-lived species of high fecundity, controlling for factors such as likelihood of exposure to specific gears);
- the likelihood that the gear will encounter the species there should be a relatively high probability of exposure to the gear);
- the likelihood that an encounter with the gear will kill or injure the species (species which are soft or brittle may be more vulnerable than species with hard shells or leathery epidermis);
- the proportion of the population which is in the area where the fishery operates (a large part of the species' range should lie within the area of activity of the fishery on macro [geographic] and micro [habitat] scales);
- it must be possible to quantify at least the sign of the trend of the population, and ideally more;
- moreover, because most population trends are likely to be affected by several factors as well as fishing (Daan *et al*., 1996), it will often be important to monitor several areas with substantial contrast in fishing intensity.

7.6.1.2 Range

For reference points addressing reduction of range and loss of population structure, the same reasoning applies with regard to the impossibility of assessing all species and the need to select species whose conservation is likely to ensure conservation of less vulnerable species. Within the field of ecology there is significant debate and conflicting data about the relationship between population size and range occupied (see MacCall, 1990; Fretwell, 1972). The current weight of evidence suggests that it is not appropriate to generalize that a reduction in range necessarily corresponds to a decline in abundance. Nonetheless, it is a symptom which warrants investigation when observed (e.g., Baltic cod, Section 4.2). The assumption that a reduction in range corresponds to a reduction in abundance may be safer for moderately sedentary species than for highly mobile ones, particularly if the mobile species routinely migrate extensively and opportunistically. Therefore, the assumption may be appropriate for many benthic species.

For the sedentary species information is usually lacking regarding the dependence of local recruitment on local spawning. Hence, there is likely to be controversy about the scale at which a documented effect should trigger a management action, that is, about the value of the precautionary reference point for range reduction. The properties characteristic of a good candidate species for setting reference points regarding range reduction vary with the mobility of the species.

For species which are moderately sedentary, appropriate properties include:

- presence and abundance can be quantified well with properly designed monitoring programmes, including the use of proper statistical approaches to analysing change in infrequent observations, if the species is uncommon;
- information linking fishing to the loss or depletion of local populations is sound—this often requires evidence of direct mortality, physical injury from gear combined with increased risk of predation, or loss of essential habitat features for the species caused by fishing gear (use of the latter type of evidence also presupposes knowledge of essential habitat for the species);
- it possesses at least some of the characteristics of vulnerability discussed under direct mortality;
- it is desirable, but not essential, that there be some knowledge of the degree to which local recruitment depends on local population status.

Even for species with these characteristics, it usually will NOT be clear what decline in range should be used as a reference point to trigger management action. Baillie and Groomsbridge (1996) and CITES (1994) have adopted range criteria, but these were developed for species with population dynamics of birds and mammals. Many sedentary benthos may be viewed more like plants, and there is substantial debate about the shape of the functional relationship linking change in range to change in abundance and threat to conservation.

If a species is quite mobile, appropriate properties include:

- factors affecting changes in distribution are known. Ideally, this includes not just knowledge of typical migration patterns, but also some understanding of how migration routes and timings, and areas occupied during a season, change with environmental conditions such as temperature, salinity, oxygen, etc.
- change in range can be documented with appropriate quantitative methods. These must reflect the uncertainty in spatial distribution appropriately, if the reference points are to have a sound relationship to degree of risk aversion.
- there should be plausible links (with some documentation) between fishing depleting local populations (the proximate mechanism could be either direct mortality or loss of essential habitat) or fishing reducing population numbers and the decline in abundance resulting in contraction of range.

Even with the above information available for a candidate species, it often will not be clear what decline in range should be used as a reference point to trigger management action. Because of at least differences in dispersal properties of reproductive propagules, criteria developed for birds and mammals may not be appropriate for mobile marine species. Present knowledge of the spatial dynamics of most mobile marine species is inadequate to state how large a decrease in range corresponds to a marked increase in likelihood that the population is suffering unsustainable mortality. Moreover the functional relationship of abundance to range is likely to be non-linear have species-specific parameters which could vary with migration habits, diets, and life history parameters, and be difficult to parameterize.

7.6.1.3 Ecologically dependent species

For **ecologically dependent species,** the same reasoning applies with regard to the need to select species whose conservation is likely to also ensure that less dependent species are not at risk from the fishery depleting the common food supply. Some ecologically dependent species (particularly seabirds and marine mammals) show parental care, so food depletion may be detected with reproductive failure rather than waiting for population-scale response to be quantified.

Characteristics for good species for which to set reference points include:

- diet is reasonably well known, including information on inter-annual variability;
- evidence is available that the species of prey being harvested by the fishery is well represented in the diet;
- evidence is available that prey-switching from the species being harvested is rare, or at least does not result in complete compensation when the prey has become rare;
- evidence is available that the foraging range of the species of interest does not extend well beyond the region of operation of fishery on a time scale relevant to the rate of renewal of the prey;
- there is a population parameter (such as breeding success, growth rate) related to feeding whose trend can be quantified. The population parameter is best if it is not strongly influenced by non-feeding conditions.

Because many population parameters are influenced by diverse environmental factors, the reference point suitable to trigger management action may have to be a sustained change in the population parameter, corresponding to activity of the fishery over a comparable period.

As an example, the ICES Study Group on Effects of Sandeel Fishing (ICES, 1999b) presents a rationale for using a three-year depression in breeding success of kittiwakes (*Rissa tridactyla*) as a reference point corresponding to local depletion of sandeels.

7.6.1.4 Scavengers

For **scavengers** one is considering management action to address a higher order relationship, in that the increase in scavenging species is only a concern because they may reach abundances where they are detrimental to other populations. Correspondingly, an appropriate reference point must be for a species whose populations are likely to be negatively impacted by abundant scavengers. Moreover, one must be confident that the scavengers presenting the threat to the species of concern are those whose populations are benefiting from fish remnants produced by the fishery. If both the scavengers and the populations that the scavengers are affecting are birds, it might be appropriate to use declining access to breeding sites, increased disturbance of breeding activities (from courtship to fledlging success), or direct mortality as indicators of impact.

Characteristics of good species for which to set reference points for impacts of scavengers have many similarities with criteria for choosing ecologically dependent species (Section 7.6.1.3), and include:

- The link between the scavenger population and the population of concern is tight, and well documented.
- The feature(s) of the population of concern which are being monitored can be quantified well.
- The trend in the feature being monitored can be shown to be causally linked to the impact of scavengers, and is not often likely to experience large perturbations due to the other factors.
- The increase in the scavenger population can be shown to be causally linked to the provision of fish remnants.

7.6.2 Habitats

Protection of habitats is a prerequisite for protecting the species dependent upon the habitats. Given the recognised loss of habitat features in some areas, development and implementation of ecosystem management objectives ensuring the protection of the remaining areas must be seen as a priority, particularly if habitat features which are vulnerable to disturbance (see Section 6.2) are uncommon.

The most straightforward approach to habitat protection is the complete exclusion of damaging activities from all habitats at risk. It may, however, be that a certain level of habitat degradation may be acceptable, for example because the effects are reversible.

Ultimately management for habitat considerations may extend to all habitats, but at least initially such considerations are likely to be restricted to a sub-set of habitats. We set out below factors, which may influence the choice of such a sub-set, and the reference points that might be appropriate.

7.6.2.1 Criteria for selection

Criteria that might be used to select habitats for conservation include:

- High degree of ‘endemic’ biota, for example, sea lochs and coastal lagoons.
- Restricted distribution, inherently rare habitats such as *Lophelia* reefs.
- High biological diversity. The Jakarta Mandate requires protection of habitats with high biological diversity—candidate areas might include sub-littoral reefs and boulder beds.
- EC Habitats and Species Directive Annex 1 list—the EC Directive provides a list of habitats within Europe which it believes should be protected.
- Identified in Biodiversity Action Plans—in the UK this includies *Sabellaria* reefs, *Modiolus* beds, *Lophelia* reefs, deep mud.
- Essential fish habitat—such as gravel banks for herring spawning.

7.6.2.2 Possible reference points

The stage of development of reference points for populations is well in advance of that for habitats. Current knowledge therefore does not allow a full discussion (cf. Section 7.5.1), rather we point to features which warrant further investigation:

- Proportion of initial area maintained in un-impacted condition.
- Some property of the spatial distribution—e.g., minimum of *n* % in un-impacted condition in any ICES rectangle. This, at least partially, addresses the issue of patchiness.
- Some measure of habitat quality (e.g., epibiota: biomass per unit area) across the whole habitat unit. Such reference points would allow some use of, and hence effect on, an area. This requires knowledge of the form of the relationship between the degree of change in range and the risk that the change is irreversible.
- As current management of target species is done within a precautionary framework, including multiple reference points (biomass and fishing mortality), management of habitats may also require combinations of criteria. Such a criterion might be ‘no more than *x* % change in a metric in the entire habitat unit and a minimum of *y* % in un-impacted condition in an ICES rectangle’.

7.6.3 Genetic properties

A number of management measures are available to conserve genetic diversity of exploited species (ICES, 1999a) and some of these could require reference points specific to genetic properties of the stock or species. Genetic diversity is directly related to N_e, the effective number of spawning individuals in a population, and the most appropriate variable for assessing population viability (Barton and Whitlock, 1997). Complex social systems, skewed sex ratios, and other complicating factors of breeding systems may result in N_e being smaller than the number of mature individuals in a population (Burgman *et al*., 1993). Maintaining large N_e increases the likelihood that favourable mutations will become widespread and deleterious ones will be unduly expressed.

Population size is the single most important factor in sustaining a high level of genetic variation within a population of a species, and for essentially all fished species maintaining a population above B_{pa} has a high likelihood of ensuring that the number of potential breeding individuals also exceeds $N_{e.}$

Given a mean population size, N_e is negatively influenced by extreme fluctuation in population size, variation in the number of offspring per family and unbalanced sex ratios. Keeping a population above B_{pa} will prevent fluctuations serious enough to result in unacceptable risk to N_e Variation in offspring per family is not amenable to measurement or control in the wild, so reference points addressing that factor usually are not appropriate. (This could be a concern for harvesting of moderately sedentary intertidal species, such as abalone (*Haliotis*) and sea urchins (*Strongylocentrotus*), where ‘mating’ opportunities are restricted by the linear nature of the habitat. However, setting B_{pa} can accommodate the need for a reasonable density as well as abundance of mature individuals.

The sex ratio of a population is rarely considered as a management objective, although if the sex ratio of breeders departs from 1:1, N_e and genetic variation will be reduced. An effective population of 50 males and 50 females is nearly 2.8 times larger, genetically, than one of 10 males and 90 females. Some jurisdictions manage species such as snow crabs (*Chionocetes*) and shrimp (*Pandalus*) with size limits which allow only males to be harvested. In such cases target exploitation rates are set to ensure that enough males survive to mate with all females. Under such approaches, it is unlikely that the skew in sex ratio will be so bad that N_e reaches values which reflect significant risk to the population.

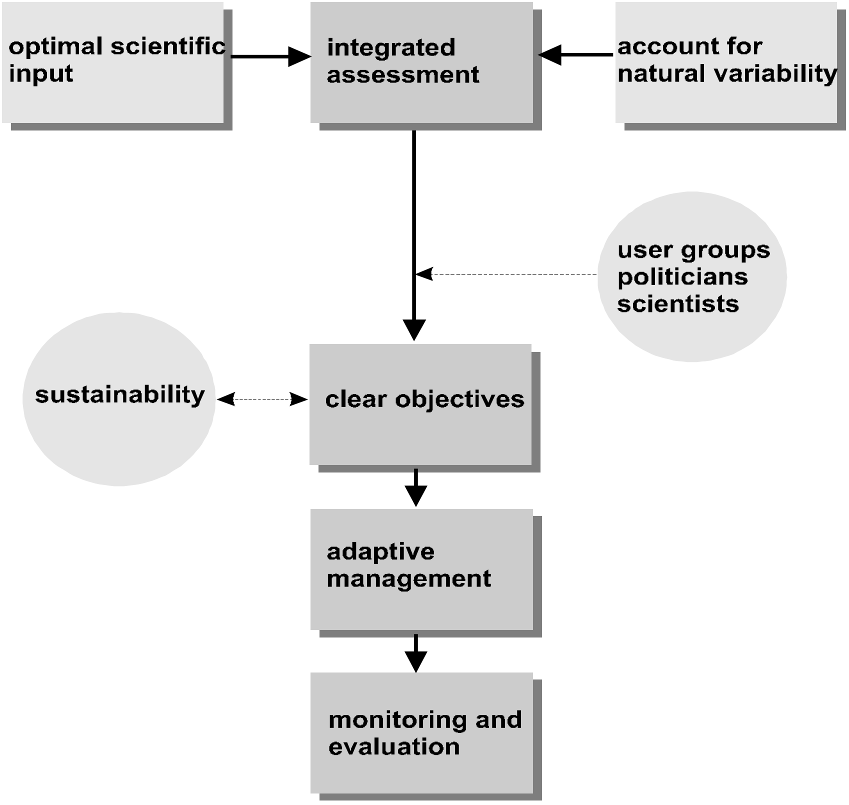
For species where there is a high degree of population sub-division, that is high among-population genetic variation, reference points may be needed for the individual populations. Tools for population risk assessment, such as population viability analysis, may be appropriate for developing reference points for subpopulations (Burgman *et al*., 1993; Beissinger and Westphal, 1998; Dunham *et al*., 1999). The reference points themselves, however, are still likely to be numbers or biomasses, and function like B_{pa}. When the extinction risk of many local populations must be considered, the same problems of practicality are encountered as with reference points for all possible species of by-catch. Suggestions in Section 7.5.1 are relevant here. In addition, Allendorf *et al*. (1997) have provided a set of qualitative criteria for ranking conservation value of salmonid stocks, and these warrant review for wider application.

Reference points for selection differentials may be important, but further work within that field is required before it will be possible to identify reference points which can be applied within existing precautionary frameworks. More must be known about the relationship between selection differential and conservation risk, and how to measure selection differentials in operational settings, before reference points can be proposed for this important property.

7.6.4 Emergent properties

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 metrics of ecosystem properties, such as measures of diversity, can serve a valuable role in communicating with many clients of marine science, for example as part of the approach proposed in Lanters (1999) and illustrated in Figure 7.6.4.1.

Figure 7.6.4.1. A framework for ecosystem management. The central line shows how to make it operational. Other elements are preconditions for ecosystem management (from Lanters, 1999).



7.7 Conclusions and Way Forward

A number of international agreements require protection of the marine ecosystem. WGECO believes that in some areas there is now urgent need for key habitats to be afforded protection. A difficulty at present is that our knowledge about benthic habitats is limited. We have some knowledge about soft bottom habitats and communities, but the diversity of habitats associated with hard bottoms and their special topographical features, including habitat-forming species such as deep-sea corals and *Sabellaria* reefs, are not well known. There is therefore a need for classifying and mapping the distribution of benthic habitats in the North Sea. WGECO, therefore, supports the recommendation of the ICES Study Group on Marine Habitat Mapping that joint efforts in habitat mapping are beneficial to the interests of ICES and WGECO. **A potentially useful Term of Reference for the future would be a consideration of the application of habitat classification and mapping (including GIS) to integrated environmental management incorporating fishing effects.**

Development and implementation of population reference points for non-target species is hampered by our lack of knowledge of the biology and ecology of many species and the often rather subjective allocation of taxa to groupings such as ‘sensitive to fishing’. There is a need to increase our knowledge of the ecology of the benthos and the development of robust and objective criteria, and scales/metrics, for the independent assessment of vulnerability/fragility of habitats and species.

It is generally accepted that discards have a negative effect on the ecosystem. They provide no economic return and the extra time spent sorting the catch places an economic burden on the industry. Minimising unwanted catch must therefore remain an important management objective. This must be achieved by better selectivity of the gear and the release back into the water of the unwanted catch alive and in good condition.

At this time WGECO believes that we are not in a position to recommend that reference points for emergent properties of ecosystems are necessary, beyond the reference points which would assure sustainability and conservation of all species and habitats impacted by fishing. Neither are we prepared to confirm that single species, habitat and genetic reference points alone are enough to ensure a precautionary approach to ecosystem management, only that no properties have been shown to be placed at risk if the constituent components are conserved.

Failure to address socio-economic issues limits our ability to make progress with implementing existing biologically based management. Further development of integrated management objectives as a basis for an ecosystem approach to management requires development of socio-economic models that allow integration of ecological and social issues.

WGECO feels that the way ahead involves:

- rapid implementation of habitat reference points for key habitats;
- rapid movement to fishing efforts that are sustainable for the target species reference points;
- further development of genetic reference points and reference points for non-target species;
- reduction in unwanted catch without increasing the quantity of damaged material left on the sea floor;
- development of mechanisms linking ecosystem management tools to appropriate reference points;
- development of strategies and tools for addressing the social costs of reduction in harvest required to meet ecosystem (and single species) reference points.

7.8 References

Allendorf, F.W., Bayles, D., Bottom, D.L., Currens, K.P., Frissell, C.A., Hankin, D., Lichatowich, J.A., Nehlsen, W., Trotter, P.C., and Williams, T.H. 1997. Prioritizing Pacific salmon stocks for conservation. Conservation Biology, 11: 140–152.

Anon, 1995. The Ecosystem Approach: Healthy Ecosystems and Sustainable Economies. Volume I. Overview. Report of the Interagency Ecosystem Management Task Force. Virginia, United States.

Baillie, J., and Groomsbridge B., 1996. IUCN Redlist of Threatened Animals. IUCN Publ. Serv. Gland, Switzerland. 368 pp.

Barton, N.H., and Whitlock, M.C. 1997. The evolution of metapopulations. *In* Metapopulation biology: ecology, genetics, and evolution. Ed. by I.A. Hanski and M.E. Gilmin. Academic Press, San Diego, California.

Beissinger, S.R., and Westphal, M.I. 1998. On the use of demographic models of population viability in endangered species management. Journal of Wildlife Management, 62: 821–841.

Burgman, M. A., Ferson, A.S., and Akçakaya, R. 1993. Risk assessment in conservation biology. Chapman and Hall, New York, NY, USA.

Christensen, N.L., Bartuska, A.M., Brown, J.H., Carpenter, S., D'Antonio, C., Francis, R., MacMahon, J.F., Noss, R.F., Parsons, D.J., Peterson, C.H., Turner, M.G., and Woomansee, R.G. 1996. The Report of the Ecological Society of America Committee on the Scientific Basis for Ecosystem Management. Ecological Applications, 6: 665–691.

CITES. 1994. Convention on International Trade in Endangered Species of Wild Fauna and Flora, Ninth Meeting of the Conference of the Parties, Fort Lauderdale (United States of America), 7 to 18 November 1994. Conference of the Parties Resolution Conf. 9.24.

Daan, N., Richardson, K., and Pope. J.G. 1996. Changes to the North Sea Ecosystem and Their Causes: Århus 1975 Revisited. ICES Journal of Marine Science, 53: 879–883.

Dayton, P.K., Thrush, S.F., Agardy, M.T., and Hofman, R.J. 1995. Environmental effects of marine fishing. Aquatic Conservation: Marine and Freshwater Ecosystems, 5: 205–232.

Dunham, J., Peacock, M., Tracy, C.R., Nielsen, J., and Vinyard, G. 1999. Assessing extinction risk: integrating genetic information. Conservation Ecology, 3(1): 2.

Fretwell, S.D. 1972. Populations in a seasonal environment. Monographs in Population Biology, 5 Princeton University Press: 217 pp.

Furness, R.W. 1998. Are industrial fisheries a threat to seabird populations? *In* Proceedings 22: International Ornithologists Congress, Durban, University of Natal. Ed. by N. Adams and R. Slotow.

Gharrett, A.J., and Smoker, W.W. 1993. Genetic components in life history traits contribute to population structure. *In* Genetic conservation of salmonid fishers, pp. 197–202. Ed. by J.G. Cloud and G.H. Thorogaard. Plenum Press, London.

Giske, J., G. Huse, and O. Fiksen. 1998. Modelling spatial dynamics of fish. Review in Fishery Biology and Fish, 8: 57–91.

Groom and Pascual. 1998.

ICES. 1996. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 1998/Assess/Env:1.

ICES. 1997. Report of the Working Group on the Application of Genetics in Fisheries and Mariculture. ICES CM 1997/F:4.

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

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

ICES. 1999a. Report of the Working Group on the Application of Genetics in Fisheries and Mariculture. ICES CM 1999/F:1.

ICES. 1999b. Report of the Study Group on Effects of Sandeel Fishing. ICES CM 1999/ACFM:19.

Lanters, R.L.P. 1999. Basic elements for the implementation of an ecosystem approach to marine management. ICES CM 1999/Z:10. 7 pp.

Law, R., and Grey, D.R. 1989. Evolution of yields from populations with age-specific cropping. Evolutionary Ecology, 3: 343–359.

Law, R., and Rowell, C.A. 1993. Cohort-structured populations, selection responses, and exploitation of the North Sea cod. *In* The exploitation of living resources, pp. 155–173. Ed. by T.K. Stokes, J.M. McGlade, and R. Law. Lecture Notes in Biomathematics 99. Springer-Verlag, Berlin, Germany. 264 pp.

MacCall, A.D. 1990. Dynamic geography of marine fish populations. University of Washington Press, Seattle, WA. 153 pp.

Mileikovsky, S.A. 1971. Types of larval development in marine bottom invertebrates, their distribution and ecological significance: a re-evaluation. Marine Biology, 10: 193–213.

Millner, R.S., and Whiting, C.L. 1996. Long-term changes in growth and population abundance of sole in the North Sea from 1940 to the present. ICES Journal of Marine Science, 53: 1185–1195.

NRC (USA). 1999. Sustaining marine fisheries. National Research Council, National Academy of Sciences, Washington, D.C. 164 pp.

Rijnsdorp, A.D. 1993. Fisheries as a large-scale experiment on life-history evolution: disentangling phenotypic and genetic effects in changes in maturation and reproduction of North Sea plaice, *Pleuronectes platessa* L. Oecologia, 96: 391–401.

Rowell, C.A. 1993. The effects of fishing on the timing of maturity in North Sea cod (*Gadus morhua* L.). *In* The exploitation of living resources, pp. 44–61 Ed. by T.K. Stokes, J.M. McGlade, and R. Law. Lecture Notes in Biomathematics 99. Springer-Verlag, Berlin, Germany. 264 pp.

Ruzzante, D.E., Taggart, C.T., Cook, D., and Goddard, S.V. 1997 Genetic differentiation between inshore and offshore Atlantic cod (*Gadus morhua*) off Newfoundland: a test, and evidence of temporal stability. Canadian Journal of Fisheries and Aquatic Sciences, 54: 2700–2708.

Sala, E.,. Boudouresque C.F, and Harmelin-Vivien M., 1998. Fishing, trophic cascades, and the structure of algal assemblages: evaluation of an old but untested paradigm. Oikos, 82: 425–439.

Shaw, P.W., Pierce, G.J., and Boyle, P.R. 1999. Subtle population structuring within a highly vagile marine invertebrate, the veined squid *Loligo forbesi*, demonstrated with microsatellite DNA markers. Molecular Ecology, 8: 407–417.

STECF. 1999. Report on the STECF Workshop on 'Research Priorities on Environmental Impact of Fishing, Fuengirola, Spain, 18–20 October 1999.

Trippel, E.A. 1998. Egg size and viability and seasonal offspring production of young Atlantic cod. Transactions of the American Fisheries Society, 127: 339–359.

Trippel, E.A., Morgan, M.J., Frechet, A., Rollet, C., Sinclair, A., Annand, C., Beanlands, D., and Brown, L. 1997. Changes in age and length at sexual maturity of northwest Atlantic cod, haddock and pollock stocks, 1972–1995. Canadian Technical Report on Fisheries and Aquatic Sciences, 2157. 120 pp.

Tuljapurkar. 1990.

van Oppen, M.J.H., Klerk, H., Olsen, J.L., and Stam, W.T. 1996. Hidden diversity in marine algae: some examples of genetic variation below the species level. Journal of the Marine Biological Association of the United Kingdom, 76: 239–242.

8 FOOD FOR THOUGHT

8.1 Social and Economic Aspects of Ecosystem Effects of Fishing

Significant progress on reducing effects of fishing on ecosystems is unlikely to be made with only the historic fisheries research community working with fisheries managers. Almost all the possible measures discussed in Section 6.4 have economic impacts, and social impacts on communities. Costs and benefits of many measures will not be distributed equally among all who will be affected by implementation of the measures. Therefore, economists and social scientists need to be brought into these discussions in many fora, including possibly some of the ICES fora. ICES has made some progress in building links to the research community on fisheries economics, and these links should be strengthened. However, WGECO thinks that many of the most difficult problems in even current single-species management of target species are more a result of the inability of managers, governments, and society to come to grips with the social impacts of necessary measures, rather than the economic consequences of those measures. WGECO believes that those problems will only become worse as measures to address effects of fishing on ecosystems are pursued. For example, societal values of protection of non-commercial species and habitats may be greatly different between communities depending on fishing and other sectors of society, but the costs of such protection may fall heavily on coastal communities. The sources and consequences of these social issues must be understood, and strategies to address them effectively must be identified. WGECO feels strongly that research on the social aspects of fisheries management and conservation must be pursued much more aggressively, and the research should progress in ways which are fully integrated with current fisheries research on effects of fishing on ecosystems**.**

8.2 Research Proposal: Biodiversity of Fish Communities—Towards Testable Hypotheses

8.2.1 Introduction

It appears that so far WGECO has only been able to identify vulnerable species *a posteriori* based on experimental results showing that a particular species has markedly decreased in abundance in response to fishing. In fact, statements about fragile and/or k-strategy species being more vulnerable than robust and/or r-strategy species are often based on circular argumentation rather than on tests of an *a priori* hypothesis. Although there is no doubt that vulnerable species will be suffering most, because the argument may be reversed by calling those species suffering most vulnerable species, this is not very helpful in predictions regarding species which have not been included in any experimental set-up. In sofar, our list of vulnerable species will always be somewhat anecdotal, because only those that happen to have been studied can be included.

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 which have been classified as the most vulnerable have actually shown a decline in response to fishing and vice versa.

The availability of a large number of research surveys as well as the general knowledge available for most fish species should allow us to carry out such a test. Moreover, because many fish species are common to several ecosystems in the North Atlantic, we could actually start to combine information from many sources and build up an expertise, which allows us to make more general predictions about the likelihood that particular species are vulnerable to fishing pressure. Although the idea is by no means new (Rochet, 1996; Jennings *et al*., 1999; Greenstreet and Rogers, 1999), pursuing the issue within the wider mandate of the remit of WGECO would be a major step forward.

The following paragraphs present a layout of the approach that might be taken within WGECO and WGEEF to do some actual analyses in this respect. Although concentrated on the IBTS database, a similar approach might be adopted for survey data from other areas. The description just serves as a start, and may be amended or extended as appropriate. However, it must be stressed that any particular survey represents a particular component of the total fish community that happens to be accessible to the gear. Although it will by no means allow a thorough analysis of the total fish fauna, the fact that the survey gear is a bottom trawling gear would make it seem likely that the data incorporate all species that might be affected by such gears.

8.2.2 Approach

Null Hypotheses

- Species at the k-end of the r/k-continuum have not shown more examples of decreased abundance over time than species at the r-end.
- Species with specific habitat requirements have not shown more examples of a reduction in geographical extension than ubiquists.
- Species richness has not decreased more in intensively fished areas than in less disturbed areas.
- More abundant species have decreased as much as less abundant species.

Data

There are three types of data required to classify the species and to measure the response:

- Classification on the r/k-continuum is not straightforward because this measure involves many aspects such as longevity, number of eggs produced, parental care, etc. These parameters may not be known for all species. However, they are strongly related to parameters such as maximum size, egg size, (ovo-)vivipary, demersal or pelagic eggs, which are probably available for most species. As a first approximation we might take a straight or somehow weighted average of the individual ranking orders for all species that have ever been caught in the survey. There may be other factors that might be used to discriminate between different classes of species such as boreal versus Lusitanian species, migrants versus local North Sea stocks, and coastal versus open sea species.
- The second source is provided by the IBTS database. The database has been updated to include all data from 1973, when the entire North Sea was first covered. By averaging hauls by statistical rectangle by year and summing over all rectangles and years, all species can be ranked according to their abundance. Moreover, the total number of squares in which each species has been recorded should reflect some measure of habitat specificity of that species allowing another ranking order. Furthermore, the total number of species recorded from each rectangle serves as some measure of local differences in diversity. The survey data can also be used to estimate trends in the various parameters over time. It should be noted that not all rectangles have been sampled in each year and the number of hauls between rectangles may vary. These statistical problems related to unbalanced sampling should be solved in order to get unbiased estimates of all parameters required.
- The third and most problematic source will be effort data by rectangle for various demersal fisheries. These will not be available as time series over the entire period, but as a first approximation the data for a particular range of years may be assumed to reflect the average difference between rectangles over the entire period.

Methods

After all sensible classifications have been finished, trends in total abundance, geographical extension and species richness will be determined and related to the different classifications in order to establish whether the Null hypotheses must be rejected. By comparing ecosystems, the generality of these findings can be ascertained. If for instance temperature data could be added, the hypothesis that high temperature would favour Lusitanian species (abundance and extension) might also be tested.

In order for progress to be made in this field, a suitable **Term of Referenc**e for the next meeting of WGECO would be:

- **to further develop testable hypotheses for evaluating which components of the marine ecosystem are most vulnerable to trawl impacts.**

8.3 References

Greenstreet, S.P.R., and Rogers, S.I. 1999. The effects of fishing on non-target fish species. *In* Effects of fishing on non-target species and habitats: biological conservation and socio-economic issues. Ed. by M.J. Kaiser and S.J. de Groot. Blackwell Science, Oxford. 416 pp.

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

Rochet, M.J. 1996. The effects of fishing on demographic traits of fishes. ICES CM/ Mini:9. 6 pp.

9 FUTURE ACTIVITIES FOR WGECO AND RELATED ICES GROUPS

9.1 Meeting / Workshop on Ecosystem Models

The goals and rationale for this proposal are presented in detail in Section 3.5. The models and analytical approaches ICES uses as a basis for single species advice undergo frequent and intensive scrutiny, as do the models used in providing multispecies fisheries advice. As ICES increases the advice provided in ecosystem contexts, it must take the same approach with ecosystem models it considers using and give them the same scrutiny. The workshop proposed in Section 3.5 is essential if ICES is to continue to expand its advisory role on ecosystem questions.

9.2 Ecosystem Effects of Deep-Water Fisheries

In Section 6, WGECO notes that effects of fishing gears on habitats are generally the most long-lasting and irreversible of all effects of fishing on ecosystems. In that context, WGECO takes note of the expansion of fisheries into new areas, as harvesting of deep-water species expands. WGECO has confidence in the approach taken by WGDEEP, in giving importance to the life history characteristics of many deep-water species, which indicate that those species can sustain only very low exploitation rates. However, WGECO would like to stress its concern about the physical and biogenic features of the deep-water habitats as well. To date, these are likely to have been much less impacted by bottom gears than habitat features on shelf seas in the ICES area, but the habitats are likely to be highly vulnerable for all the reasons described in Sections 6 and 7. WGECO urges ICES to attach priority in its advice, and management agencies in their regulations, to ensuring that these new and expanding fisheries are kept sustainable both with regard to the mortalities inflicted on all species (target and non-target) and effects on habitats.

9.3 Testable Hypotheses about Fishing Effects on Biodiversity

The rationale and possible Term of Reference for WGECO are presented in Section 8.2. We copy the Term of Reference here.

- **to further develop testable hypotheses for evaluating which components of the marine ecosystem are most vulnerable to trawl impacts.**

9.4 Liaise with the Study Group on Marine Habitat Mapping

Sections 6 and 7 of this report call attention frequently to the importance of evaluating effects of fisheries on habitats. Such evaluations can be done in only limited ways without good data on marine habitats. For that reason, WGECO encourages the newly formed Study Group on Marine Habitat Mapping to continue its efforts in habitat mapping in the ICES area. To ensure maximum usefulness of the habitat maps for evaluating the effects of fishing on marine ecosystems, WGECO **recommends that the Study Group on Marine Habitat Mapping (SGMHM) work closely with the Working Group on Ecosystem Effects of Fishing Activities, to ensure that the habitat classification system is adequate for assessing the impacts of bottom gears on benthic communities and habitats.** This collaboration has to proceed quickly, if input from WGECO is to be received in time to be of maximum value to SGMHM. **WGECO could work intersessionally to collate comments on the documents which describe the classification system and detailed approach to be taken by SGMHM, and make these comments available to the Chair of SGMHM in advance of adoption of their final decisions** on these matters. The interaction would not end there, of course, as frequent interaction might be warranted, and it would be helpful to structure an annual exchange between the two Working Groups on the Terms of Reference assigned by ICES to each group. Section 7 also proposes a Term of Reference for WGECO to:

- consider the **application** of habitat classification and mapping (including GIS) to integrated environmental management incorporating fishing effects.

which will be of interest to SGMHM as well.

9.5 Quantifying and Controlling Effort

This report and past reports of WGECO have stressed the crucial importance of controlling fishing effort if the effects of fishing on ecosystems is to be reduced. Advances in technology present an opportunity to monitor the position and activity of all fishing vessels operating in the ICES area. With the addition of strict regulation of the type of fishing gear in use on each vessel and strengthening of the current fisheries logbook and on-board observer schemes, direct quantification and monitoring of fishing effort is now possible. WGECO would like to see such initiatives pursued vigorously by all fisheries management agencies, nationally and internationally, for two reasons. One is that the availability of effort data, including information on time, place, fleet, and gear, would increase greatly our ability to measure and understand the effects of fishing on ecosystems. The other is that many of measures which could reduce the effects described in Section 6 will work much better if science and management both have knowledge of how effort is distributed in space, time, and by fleet, and how regulations affect that distribution.

ANNEX 1: LIST OF PARTICIPANTS

Name	Address	Telephone	FAX	E-mail
Magda Bergman	Netherlands Institute for Sea Research (NIOZ) PB 59 NL-1700 AB Den Burg Texel Netherlands	+31 222 369300	+31 222 31967	magda@nioz.nl
Jean Boucher	IFREMER Pêche Aquaculture BP 70 F-29280 Plouzané France	+33 2982 24615	+33 29822 4653	jboucher@ifremer.fr
Jeremy Collie	Danish Institute for Fisheries Research Charlottenlund Slot DK-2920 Charlottenlund Denmark	+45 3396 3357	+45 3396 3333	jsc@dfu.min.dk
Niels Daan	RIVO Netherlands Institute for Fisheries Research P.O. Box 68 NL-1970 AB IJmuiden Netherlands	+31 2555 6464	+31 2555 64644	niels@rivo.dlo.nl
Martin Dorn	Alaska Fisheries Science Center NMFS/NOAA 7600 Sandpoint Way NE Seattle, WA 98115–4176 United States	+1 206 526 6548	+1 206 526 6723	martin.dorn@noaa.gov
Siegfried Ehrich	Bundesforschungsanstalt für Fischerei Palmaille 9 D-22767 Hamburg Germany	+49 403 890 5179	+49 403 890 5263	ehrich.ish@bfa-fisch.de
Arill Engås	Institute of Marine Research P.O. Box 1870, Nordnes N-5317 Bergen Norway	+47 5523 6808	+47 5523 6830	arill.engas@imr.no
Dick Ferro	FRS Marine Laboratory P.O. Box 101 Victoria Road Aberdeen AB11 9DB United Kingdom	+44 1224 876544	+44 1224 295511	ferro@marlab.ac.uk
Alain Fréchet	Maurice Lamontagne Institute P.O. Box 1000 Mont-Joli, G5H 3Z4 Québec Canada	+418 775-0628	+418 775-0679	frecheta@dfo-mpo.gc.ca
Chris Frid	Dove Marine Laboratory University of Newcastle Cullercoats North Shields NE30 4PZ United Kingdom	+44 191 252 4850	+44 191 252 1054	c.l.j.frid@ncl.ac.uk
Simon Greenstreet	FRS Marine Laboratory P.O. Box 101, Victoria Road Aberdeen AB11 9DB United Kingdom	+44 1224 295417	+44 1224 295511	greenstreet@marlab.ac.uk

Name	**Address**	**Telephone**	**FAX**	**E-mail**
Sture Hansson	Department of Systems Ecology Stockholm University S-10691 Stockholm Sweden	+46 816 4248	+46 815 8417	sture.hansson@system.ecology.su.se
Louize Hill	IPIMAR Avenida de Brasilia P-1400 Lisbon Portugal	+351 1301 7062	+351 21301 5948	lhill@ipimar.pt
Erik Hoffmann	Danish Institute for Fisheries Research Charlottenlund Slot DK-2920 Charlottenlund Denmark	+45 3396 3300	+45 3396 3333	eh@dfu.min.dk
Ellen Kenchington	Bedford Institute of Oceanography PO Box 1006 Dartmouth, NS B2Y 4A2 Canada	+1 902 426 2030	+1 902 426 1862	knechington@mar.dfo-mpo.gc.ca
Knut Korsbrekke	Institute of Marine Research P.O. Box 1870, Nordnes N-5317 Bergen Norway	+47 5523 8500 +47 5523 8638	+47 5523 8687	knutk@imr.no
Ronald Lanters	National Institute for Coastal and Marine Management Postbox 20907 NL-2500 EX Den Haag Netherlands	+31 70 311 4324	+31 70 311 4300	r.lanters@rikz.rws.minvenw.nl
Brian MacKenzie	Danish Institute for Fisheries Research Charlottenlund Slot DK-2920 Charlottenlund Denmark	+45 3396 3300	+45 3396 3333	
Philip MacMullen	Seafish Standren’s Dock Hull, HU3 4QE United Kingdom	+44 1482 327837	+44 1482 587013	p_macmullen@seafish.co.uk
Eyfinn Magnussen	Fisheries Laboratory of the Faroes Faroe Islands	+298 315304	+298 316844	eymag@nvd.fo
Livingston Marshall, Jr.	Biology Department Morgan State University Baltimore, MD 21251 United States	+1 443 885 4468	+1 410 426 4732	lmarshall@moac.morgan.edu
Stefan Áki Ragnarsson	Marine Research Institute Skúlagata 4 121 Reykjavík Iceland	+354 552 0240	+354 562 3790	steara@hafro.is
Jake Rice *Chair*	Canadian Stock Assessment Secretariat Fisheries Research Branch – DFO 200 Kent Street Ottawa, Ont K1A OE6 Canada	+1 613 990 0288	+1 613 954 0807	ricej@dfo-mpo.gc.ca
Stuart Rogers	CEFAS Pakefield Road Lowestoft Suffolk NR33 0HT United Kingdom	+44 1502 562 244	+44 1502 513 865	s.i.rogers@cefas.co.uk

Name	Address	Telephone	FAX	E-mail
Francisco Sanchez	Instituto Español de Oceanografía Centro Oceanográfico de Vigo Apdo 1552 36280 Vigo Spain	+34 986 492 111	+34 986 492 351	fsanchez@st.ieo.es
Mark Tasker	Joint Nature Conservation Committee Dunnet House 7, Thistle Place Aberdeen AB10 1UZ Scotland United Kingdom	+44 1224 655701	+44 1224 621488	tasker_m@jncc.gov.uk mltasker@aol.com

ANNEX 2: LIST OF WORKING DOCUMENTS

WGECO 1999/1	ICES. 1998. Report of the Working Group on Ecosystem Effects of Fishing Activities, pp. 169–197, 221–231, 253–254. ICES CM 1998/ACFM/ACME:1.
WGECO 1999/2	ICES. 1998. Report of the Advisory Committee on the Marine Environment, pp. 161–184. ICES Cooperative Research Report No. 233.
WGECO 1999/3	ICES. 1997. Report of the Advisory Committee on the Marine Environment, pp. 91–95. ICES Cooperative Research Report No. 222.
WGECO 1999/4	ICES. 1997. Report of the Study Group on Unaccounted Mortality in Fishes. ICES CM 1997/B:1.
WGECO 1999/5	Walther, Y. Bycatches of cod in Swedish trawl fishery for pelagic species in the Baltic Sea. A summarized report for the Polish-Swedish Symposium on Baltic Cod.
WGECO 1999/6	By-catch of seals
WGECO 1999/7	Lunneryd, S.G., and Westerberg, H. 1997. By-catch of, and gear damages by, grey seals (*Halichoerus grypus*) in Swedish waters. ICES CM 1997/Q:11. 10 pp.
WGECO 1999/8	Rijnsdorp, A.D., and Piet, G.J. 1999. Effect of beam trawl fisheries on the North Sea benthic ecosystem. 7 pp.
WGECO 1999/9	Lanters, R.L.P. 1999. Basic elements for the implementation of an ecosystem approach in marine management. ICES CM 1999/Z:10. 8 pp.
WGECO 1999/10	Sánchez, F., and Olaso, I. 1999. Fisheries impacts in the Cantabrian Sea using a mass-balance model. 19 pp.
WGECO 1999/11	Pope, J., and Symes, D. An ecosystem-based approach to the Common Fisheries Policy: Defining the goals. (Draft report.) 43 pp.
WGECO 1999/12	Hansson, S. 1999. Ecological aspects on fisheries management. ICES CM 1999/Z:04. 5 pp.
WGECO 1999/13	Rice, J. 1999. How complex should operational ecosystem objectives be? ICES CM 1999/Z:07. 14 pp.
WGECO 1999/14	MacMullen, P. 1999. Working Group on Fishing Technology and Fish Behaviour. (Presentation/21 slides.)
WGECO 1999/15	Lindeboom, H.J., and de Groot, S.J. (Eds.) 1998. IMPACT II. The effects of different types of fisheries on the North Sea and Irish Sea benthic ecosystem. NIOZ-Rapport 1998-1, RIVO-DLO Report C003/98. 404 pp.