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Report of the ICES Advisory Committee on the Marine Environment, 2000

Copenhagen, 26 January–2 February 2000
Copenhagen, 5–10 June 2000

International Council for the Exploration of the Sea

Conseil International pour l'Exploration de la Mer

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ICES ADVISORY COMMITTEE ON THE MARINE ENVIRONMENT

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26 January–2 February 2000

Participant	Affiliation
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Dr J. Rice[3]	Chair, WGECO

[1]Unable to attend. [2]Attended part-time. [3]Invited expert, attended part-time.

All Member Countries were represented at the meeting except Finland. Representation for Belgium was shared between Dr Cooreman and Dr Redant. Representation for Iceland was shared between Dr Steingrímsson and Dr Eiríksson.

***Secretariat members participating in the meeting or portions thereof*:**

Dr J. Nørrevang Jensen	ICES Environmental Data Scientist
Ms M. Karlson	ICES Departmental Secretary

ICES ADVISORY COMMITTEE ON THE MARINE ENVIRONMENT

LIST OF MEMBERS

5–10 June 2000

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[1]Unable to attend. [2]Attended part-time. [3]Invited expert, attended part-time.

All Member Countries were represented at the meeting except the Netherlands. Representation for Denmark was shared between Dr Pedersen and Mr Mellergaard.

Secretariat members participating in the meeting or portions thereof:

Dr J. Nørrevang Jensen	ICES Environmental Data Scientist
Ms M. Karlson	ICES Departmental Secretary

EXECUTIVE SUMMARY

The ICES Advisory Committee on the Marine Environment (ACME) met at ICES Headquarters in Copenhagen twice in 2000, with the first meeting from 26 January to 2 February 2000 and the second from 5 to 10 June 2000. At the first of these meetings, the ACME conducted a scientific peer review of the OSPAR Quality Status Report 2000, prepared a report on the "Status of Fisheries and Related Environment of Northern Seas" for the Nordic Council of Ministers, and prepared a response to a request from the European Commission DG FISH. At the second meeting, the ACME prepared responses to the other requests made to ICES by the OSPAR Commission and the requests from the Helsinki Commission. This report contains these responses. In addition to responses to direct requests, this report summarizes the deliberations of ACME on topics for which advice was not directly requested but for which the ACME felt that there was information that would be of potential interest to the Commissions, ICES Member Countries, and other readers of this report.

Information in direct response to requests from, or which is relevant to, the work of both the OSPAR Commission and the Helsinki Commission

Monitoring

In 2000, the ACME continued work on the development of biological effects monitoring programmes. The ACME considered the influence of fluctuations in salinity on biomarker and bioassay responses of marine organisms (Section 6.1.1) and the use of *in situ* bioassays for evaluating effects of contaminants (Section 6.1.2). In addition, the results of a study of the influence of natural factors on biomarker responses have been reviewed (Section 6.6.4).

In continuation of previous work to identify which contaminants can be monitored on a routine basis with adequate interlaboratory comparability, the lowest concentrations of trace elements that can be monitored in sea water, sediments, and biota are listed in Section 6.4.

The issue of normalization techniques for contaminant concentrations in sediments was considered in detail. Despite the large effort devoted to this topic by a number of scientists, it has still not been possible to reach full agreement on guidelines for the normalization of contaminants in sediments. However, information on this topic is contained in Section 6.5.1, and a more detailed discussion of the strengths and weaknesses of a number of approaches to normalization is contained in Annex 1 of this report.

The ACME reviewed new information on statistical considerations relative to monitoring programmes (Section 6.6). Statistical methods for designing sampling allocation strategies and assessing monitoring programmes are discussed in Section 6.6.2. In addition, building on material from the 1999 ACME report, further material is presented with regard to determining the number of samples of sediment or biota to characterize an area in monitoring studies (Section 6.6.3).

The ACME discussed the problem of evaluating the effectiveness of monitoring programmes in determining trends against a background of natural fluctuations (Section 6.7) and concluded that a workshop is needed to consider this topic in adequate detail.

Finally, the ACME took note of guidelines for monitoring biological parameters in the marine environment (Section 6.8), particularly the recent publication of the *ICES Zooplankton Methodology Manual.*

Quality Assurance and Intercomparison Exercises

The ACME reviewed the results of quality assurance-related activities conducted during the past year and provided summaries of this work in Section 7 of this report.

In relation to quality assurance of biological measurements in the Baltic Sea, the ACME reviewed the results of the work on this topic during the past year (Section 7.1). In particular, the conduct of taxonomic training courses for phytoplankton and macrozoobenthos and the updating of taxonomic checklists were reviewed. Initial consideration was also given to the development of quality criteria for the use of biological data in assessment activities; this issue will be of high priority in the future work of the group. The ACME stresses the importance of all countries around the Baltic Sea participating in the QA work for biological measurements, to improve the comparability of measurements in this area, particularly for the HELCOM COMBINE Programme.

For the OSPAR area, the ACME noted that principles have now been defined for good practice for the sampling and analysis of chlorophyll *a*, phytoplankton, and imaging techniques (Section 7.2). Work has also continued on the development of draft guidelines for quality assurance (QA) of biological measurements under OSPAR, including the general QA system in relation to survey objectives and design, as well as more detailed QA guidance for every step in sample treatment from sampling to data handling. The detailed guidelines will cover the specific parameters to be monitored, i.e., chlorophyll *a*, phytoplankton, macrozoobenthos, and macrophytobenthos. The ACME again stresses the importance of all countries participating in the OSPAR monitoring programme for biological community parameters to participate in this QA work.

A review of the considerable amount of work to develop quality assurance procedures for biological effects techniques, including fish diseases, is contained in Section 7.3. Most of this work is being conducted under the EU-funded project Biological Effects Quality Assurance in Marine Monitoring (BEQUALM).

With regard to chemical measurements, further progress has been made in the development of additional technical annexes for the "Guidelines on quality assurance of chemical measurements in the Baltic Sea", that were initially prepared in 1997 for the monitoring programmes carried out under the Helsinki Commission (Section 7.4). Technical notes have now been completed on a) units and conversions; b) the determination of salinity and temperature; and c) the determination of co-factors. The ACME noted the success of the Second ICES/HELCOM Workshop on Quality Assurance of Chemical Procedures, which covered QA for chemical measurements in association with both environmental and input monitoring programmes.

Given the importance of certified reference materials (CRMs) for the QA of marine monitoring programmes, the ACME has included a list of relevant CRMs in this report (Section 7.5 and Annex 4).

Contaminants in the Marine Environment

The ACME reviewed brief notes on volatile organic contaminants in biota and on *tris*(4-chlorophenyl)methanol (TCPM) and *tris*(4-chlorophenyl)methane (TCPMe) in marine mammals (Section 8.1). In addition, the issue of dioxins in the marine environment was considered, particularly in relation to a Norwegian study of the abiotic and biotic dynamics and distribution of dioxins in Frierfjord (Section 8.2).

The ACME considered the results of a Norwegian study on the relationship between soft-bottom macrofauna and PAHs from smelter discharges in Norwegian fjords and coastal waters (Section 8.3). This study measured an entire species assemblage in relation to PAH concentrations in sediments, together with other variables as explanatory and environmental variables in combination with appropriate statistical tools. This approach was found to be more sensitive than previously used approaches based on diversity indices to demonstrate effects of PAHs on soft-bottom macrofauna.

A review of the types of bioassays used in the assessment of toxicity of dredged marine materials in several ICES Member Countries is contained in Section 8.4. The ACME concluded, *inter alia*, that the use of a suite of bioassays is needed owing to the differences in ecotoxicological responses to chemicals among species. In addition to the use of bioassays, there is a need to assess the overall impacts at disposal sites. There is also a need to account for the presence and the effects of persistent and/or bioaccumulative substances and to develop the use of biomarkers to increase sensitivity and contaminant-specificity.

The topic of endocrine disruption in marine and estuarine species continues to be of high priority and there is a need to develop robust techniques for the detection of endocrine disrupting chemicals and their effects in a variety of organisms. Some new techniques in pathology for the detection of endocrine disrupting chemicals in marine and estuarine organisms are described in Section 9.6, while the issue of potential genetic implications of endocrine disruption are considered in Section 15.2.

Report sections responding to requests specific to the OSPAR Commission

Trend Assessment Tools for Data on Inputs of Contaminants

A continuation of the response to the very detailed OSPAR request for advice on trend assessment tools for input data is contained in Section 6.6.1 and Annex 2. This comprises further responses to requests concerning fine tuning of the statistical method used by OSPAR for inputs (termed "Trend-y-tector"), including advice on robust smoother methods, the adjustment of loads, the analysis of monthly data, and the use of annual mean concentrations. Further guidance is also provided for reviewing the Trend-y-tector and preparing guidelines for its use.

VIC Programme on Monitoring of Temporal Trends in Contaminants in Fish

The ACME reviewed the results of the Voluntary International Contaminant (VIC) Programme on monitoring temporal trends in contaminants in fish, which was designed to provide information on small-scale temporal and spatial variations, in an effort to improve the efficiency of the programme. Unfortunately, the data sets provided for this programme were very patchy with regard to species, geographical and temporal scales, and sampling design. Nonetheless, the results so far indicate that, at least for some species and contaminants, the within-year variance could be substantially reduced by sampling from more than one site or repeating the sampling over time. This information is summarized in Section 6.6.5.

Data Handling

The annual review of data handling activities by the ICES Environmental Data Centre relevant to the requirements of OSPAR, HELCOM, and AMAP is contained in Section 20.1 of this report. Section 20.2 summarizes the work of the ICES Oceanographic Data Centre in handling nutrients data relevant to the OSPAR programmes. A brief review of the development of the biological community data reporting format and databases for phytoplankton, zooplankton, phytobenthos, and zoobenthos is given in Section 20.3. Finally, initial progress on the development of reporting formats for data obtained using the biological effects techniques adopted by OSPAR is noted in Section 20.5, with some details in Annex 10.

<u>Report sections responding to requests specific to the Helsinki Commission</u>

Triennial Review of Populations of Marine Mammals in the Baltic Sea

A summary of the status of the populations of the three species of seals and the one species of cetaceans in the Baltic Sea area is provided in Section 13.1, including also the effects of contaminants, the reproductive capacity, and the health status of these populations. To the extent possible, estimates have also been provided of the by-catch of marine mammals in Baltic Sea fisheries. ICES notes that the abundance of the harbour porpoise in the Baltic Sea area is unknown but assumed to be at very low levels, and any by-catch may pose a threat to the viability of this species in the Baltic Sea. Therefore, ICES recommends that systems be established for obtaining estimates of the total by-catch of marine mammals in Baltic Sea fisheries, and that relevant by-catch mitigation measures be implemented with particular emphasis on localized areas where by-catches are assumed to be high.

Workshop on Background/Reference Values for Concentrations of Nutrients and Chemical Contaminants in the Baltic Sea

The ACME reviewed the history of work conducted on the derivation of background/reference values for concentrations of nutrients and chemical contaminants in marine media and recalled previous ICES advice on this topic. Specific advice for use at the HELCOM Workshop is contained in Section 6.2. The ACME, however, feels that further development of assessment tools for concentrations of nutrients and chemical contaminants in the marine environment is necessary.

Effects of the Disposal of Fish Offal and Discards in the Baltic Sea

The ACME considered estimates prepared of the quantities of discards from commercial fisheries in the Baltic Sea and of the offal discharged in these fisheries together with estimates of the quantities of discards and offal consumed by seabirds. Excluding industrial fisheries, where no discarding normally is practiced, the overall discard rate in commercial fisheries in the Baltic is 3.8 % by weight, which is small compared to many other sea areas. Over half of this amount is estimated to be consumed by seabirds. With regard to offal, it is estimated that 97 % is consumed by seabirds. These estimates are contained in Section 11, with details concerning the consumption by seabirds in Annex 7.

Chapter on "Marine Fish Migratory and Freshwater Species in the Baltic Sea Area" for the Fourth Periodic Assessment of the State of the Baltic Marine Environment, 1994–1998

Section 18.2 describes the work conducted by ICES to prepare this chapter for HELCOM. This chapter was completed by the ACME and delivered to HELCOM in June 2000. Because the material provided by the various ICES Working Groups as contributions to this chapter far exceeded the page allocation for the chapter assigned by HELCOM, the ACME decided to include some of this detailed material in various sections of this report.

Report section responding to requests specific to the European Commission DG FISH

ICES prepared material in response to a DG FISH request to consider the report "The effects of different types of fisheries on the North Sea and Irish Sea benthic ecosystems" and to formulate management advice as to how the effects of the gears discussed in the report on benthic ecosystems could be measurably reduced without unduly reducing the possibilities of catching commercially important species. The review of this report and advice on relevant management measures to mitigate the effects identified in the report are contained in Section 5.

Information on topics of general interest

Ecosystem Effects of Fishing

The ACME considered a review of the ecosystem effects of fishing in the Baltic Sea that was prepared as a contribution to the chapter on fish for the HELCOM Fourth Periodic Assessment of the Baltic Marine Environment, 1994–1998. The full material is contained in Section 18.1, while a summary was provided to HELCOM.

In addition, Section 18.5 contains a review of the principal models of ecosystem dynamics; however, much of the comparative testing and critical evaluation of all models still needs to be done.

Fish Diseases

An overview of new trends in the occurrence of diseases in wild and farmed fish and shellfish stocks is contained in Section 9.1. The *paramoeba*-associated mortalities of lobsters in New York and Connecticut waters was considered to be particularly serious, although the cause of this disease is not completely clear.

Further work on the statistical analyses of disease prevalence data for dab and flounder was reviewed (Section 9.2). This work concentrated on the development of a new statistical approach to incorporate interpolated values for environmental and fisheries data in the investigation of possible relationships between environmental factors and fish diseases. On the basis of this work, ICES again encourages Member Countries to enhance their efforts to submit historic and current data held in national data banks to the ICES data banks, to facilitate a more comprehensive analysis of the interactions between natural and anthropogenic environmental factors and the health status of marine organisms.

In association with the preparation of 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 Baltic Marine Environment, 1994–1998 (Section 18.2), a comprehensive report on diseases and parasites of fish in the Baltic Sea was prepared (Section 9.4). The full report can be found in Annex 6.

Effects of Extraction of Marine Sand and Gravel on Marine Ecosystems

The ACME reviewed approaches to the assessment of the environmental effects of marine extraction activities in ICES Member Countries, as summarized in Section 16.1. The biological effects of dredging in the marine environment are considered in Section 16.2, covering effects on seabed communities, finfish, shellfish, and primary production. The nature of the biological effects is also considered in this section.

Introductions and Transfers of Marine Organisms

The ACME reviewed information on the status of accidental introductions and transfers of non-native marine species into the waters of ICES Member Countries (Section 10.1). A growing number of accidental transfers have now been documented, however, it is not clear which of these accidental introductions are of major ecological significance. The ACME has requested an assessment of the long-term implications of these introductions to be conducted.

Issues relevant to the transfer of organisms via ships' ballast water and sediments are reviewed in Section 10.2. This material shows the need for a detailed overview of the status of biological and ecological research on ballast water and sediments, and on the development of ballast water control and management technologies.

Seabird Issues

The ACME considered information on the effects of contaminants on seabirds and appropriate methods for monitoring these effects. This information is summarized in Section 12.1. Section 12.2 provides a brief summary of the effects of fisheries on seabird communities. This shows that such effects can be both negative as well as positive, with the positive side including the provision by fisheries of discards and offal as an extra source of food. Detailed estimates of the consumption of discards and offal by seabirds in the Baltic Sea are contained in Annex 7.

Issues Related to Mariculture

The ACME took note of monitoring programmes in ICES Member Countries for environmental impacts of finfish mariculture facilities, briefly summarized in Section 14.1.1. The need for monitoring programmes to evaluate the impact of shellfish aquaculture on the environment has now been identified, and the ACME agreed that the focus of this monitoring should be on the benthos. The ACME continued its consideration of the control of sea lice in salmon cultivation (Section 14.1.2), both in terms of the chemicals authorized for treating sea lice and the strategies for such treatment, including the number of lice to trigger mandatory treatment.

Information is also provided on mariculture activities in the Baltic Sea (Section 14.2), which was collected for the preparation of a chapter for the HELCOM Fourth Periodic Assessment of the State of the Baltic Marine Environment.

Issues Related to Genetics

The ACME considered the genetic effects of the restocking programmes for Baltic salmon that have been conducted for a number of years (Section 15.1). The evidence shows that some genetic changes, albeit not substantial, have occurred in Baltic salmon, and that there is no return to the original state of Baltic salmon populations. Thus, conservation of genetic diversity should be planned from the present situation. ICES has provided a series of recommendations for action with the aim of maintaining genetic diversity in Baltic salmon.

Some research recommendations have also been provided to study the potential genetic implications of commercial fisheries on deep-water fish stocks (Section 15.3).

Marine Habitat Classification and Mapping

The ACME reviewed and commented on recent developments in marine habitat classification systems and their potential application to the ICES area as a whole (Section 17). This section also contains a brief review of habitat mapping programmes in the ICES area. The ACME agreed that habitat classification and mapping should be approached initially on a large scale, subsequently working down to finer scales.

ICES Environmental Status Report

Contributions to the ICES Environmental Status Report for 2000 have been made concerning oceanographic conditions (Section 8.3.1), harmful algal blooms (Section 18.3.2 and http://www.ices.ck/status/decadal/), and fish and shellfish disease prevalence (Section 18.3.3 and Annex 8).

Global Programmes

The ACME reviewed recent activities by ICES for the North Atlantic in relation to the Global Ocean Ecosystem Dynamics (GLOBEC) programme (Section 19.1). Progress in the development of potential contributions of ICES to the Global Ocean Observing System (GOOS) were also considered, and it was noted that there are several ICES initiatives that are of direct relevance to GOOS (Section 19.2).

Sources of Information Considered by the ACME at its 2000 Meeting

At its 2000 meeting, the ACME considered, *inter alia*, information included in the most recent reports of the following ICES groups:

BEWG	Benthos Ecology Working Group
MCWG	Marine Chemistry Working Group
SGBOSV*	ICES/IOC/IMO Study Group on Ballast and Other Ship Vectors
SGDIB	Study Group on Estimation of the Annual Amount of Discards and Fish Offal in the Baltic Sea
SGEAM	Study Group on Ecosystem Assessment and Monitoring
SGMHM	Study Group on Marine Habitat Mapping
SGPHYT	Study Group on an ICES/IOC Checklist of Phytoplankton
SGQAB*	ICES/HELCOM Steering Group on Quality Assurance of Biological Measurements in the Baltic Sea
SGQAC*	ICES/HELCOM Steering Group on Quality Assurance of Chemical Measurements in the Baltic Sea
SGQAE*	ICES/OSPAR Steering Group on Quality Assurance of Biological Measurements related to Eutrophication Effects
WGAGFM	Working Group on the Application of Genetics in Fisheries and Mariculture
WGBAST	Baltic Salmon and Trout Assessment Working Group
WGBEC	Working Group on Biological Effects of Contaminants
WGECO*	Working Group on Ecosystem Effects of Fishing Activities
WGEIM	Working Group on Environmental Interactions of Mariculture
WGEXT	Working Group on the Effects of Extraction of Marine Sediments on the Marine Ecosystem
WGHABD	ICES/IOC Working Group on Harmful Algal Bloom Dynamics
WGITMO*	Working Group on Introductions and Transfers of Marine Organisms
WGMMHA	Working Group on Marine Mammal Habitats
WGMMPD	Working Group on Marine Mammal Population Dynamics and Trophic Interactions
WGMS	Working Group on Marine Sediments in Relation to Pollution
WGPDMO	Working Group on Pathology and Diseases of Marine Organisms
WGPE	Working Group on Phytoplankton Ecology
WGSAEM	Working Group on Statistical Aspects of Environmental Monitoring
WGSE	Working Group on Seabird Ecology
WGSSO	Working Group on Shelf Seas Oceanography
WGZE	Working Group on Zooplankton Ecology

Reports of the following other activities were also considered:

WKQAC*	Second ICES/HELCOM Workshop on Quality Assurance of Chemical Procedures for the COMBINE and PLC-4 Programmes

*These groups report directly to ACME.

1 INTRODUCTION

The Advisory Committee on the Marine Environment (ACME) is the Council's official body for the provision of scientific advice and information on the marine environment, including marine contamination, as may be requested by ICES Member Countries, other bodies within ICES, relevant regulatory Commissions, and other organizations. In addition, at the 1998 Annual Science Conference, the Council decided that ACME would handle all advisory tasks other than the standard fishery advisory requests, which are handled by the Advisory Committee on Fishery Management (ACFM). However, the ACFM will review fisheries-related ecosystem advice before it is sent to clients.

In handling the requests, the ACME draws on the expertise of its own members and on the work of various expert ICES Working Groups and Study Groups. The ACME considers the reports of these groups and requests them to carry out specific activities or to provide information on specific topics.

The ACME report is structured in terms of the topics covered at the ACME meeting on which it has prepared scientific information and advice; the topics include both those for which information has been requested by the Commissions or other bodies and those identified by the ACME to enhance the understanding of the marine environment. Information relevant to the Commissions' requests and specific issues highlighted by the ACME for their attention are summarized in Section 2 for the OSPAR Commission and Section 3 for the Helsinki Commission, where the individual work items from each Commission are listed and related to relevant sections of the main text.

In 2000, in addition to the usual Spring meeting of the ACME, an extraordinary meeting was held from 26 January to 2 February. Three topics were handled at this meeting: (1) a scientific peer review of the OSPAR QSR 2000 (see Section 3.2), (2) finalization of a report entitled "Status of Fisheries and Related Environment of Northern Seas" for the Nordic Council of Ministers (Section 4), and (3) a request from EC DG FISH to consider the report "The effects of different types of fisheries on the North Sea and Irish Sea benthic ecosystems" and formulate management advice to reduce the effects of bottom gears. The full advice regarding this latter request is contained in Section 5 of this report.

2 PROGRESS ON TASKS FOR THE OSPAR COMMISSION, INCLUDING DATA HANDLING

2.1 Progress on the ICES Work Programme for 2000

A summary of the progress on the 2000 programme of work requested by the OSPAR Commission is given below, along with reference to the relevant sections and annexes of this report where more detailed information can be found. This summary is provided according to the format of the Work Programme, with the questions on the Work Programme shown in italics and a summary of the ICES advice below in normal print.

SCIENTIFIC ADVICE

1 QUALITY ASSURANCE

1.1 To continue to operate the joint ICES/OSPAR Steering Group on Quality Assurance of Biological Measurements related to Eutrophication Parameters (chlorophyll-a, phytoplankton, macrozoobenthos and macrophytobenthos) in order to coordinate:

a. the development of quality assurance procedures;

b. the implementation of quality assurance activities, e.g., the conduct of workshops and intercomparison exercises;

c. the preparation of appropriate taxonomic lists of species.

Information on progress in the work of the ICES/OSPAR Steering Group on Quality Assurance of Biological Measurements related to Eutrophication Effects (SGQAE) is contained in Section 7.2 of this report. SGQAE has prepared draft general guidelines on quality assurance for biological monitoring in the OSPAR area. These guidelines, in addition to the recommendation of general procedures, encompass every step in sample treatment from sampling to data handling. These draft guidelines will be reviewed intersessionally and adopted in 2001.

2 COMPLETION OF WORK ON ASSESSMENT TOOLS

2.1 Fine tuning of the Trend-Y-tector for trend detection in inputs

2.1.1 Consider type and specification of the Smoother. The current LOESS Smoother seems to be quite sensitive for outliers in the first and last part of the data series. ICES is asked to consider possible alternatives, e.g., the Spline Smoother.

2.1.2 Consider the calculation of residuals. Due to the risk that over-fitting of normal residuals may lead to underestimation of the standard deviation, ICES is asked if residuals based on cross validation may be a sensible alternative.

2.1.3 Consider the calculation of standard deviation. The current guidelines use the L moments for estimating the standard deviation. ICES is asked to consider the statistical consequences of using this or other robust estimators combined with corresponding critical values calculated by simulation studies.

2.1.4 Examine alternatives of the current smoother test. This test does not seem to be accurate. ICES is asked for possible alternatives, e.g., splitting the trend into a linear and a non-linear component and then testing the linear part.

2.1.5 Establish the link between overall significance and individual significance levels.

2.1.6 Specify the procedure for power calculation, and especially the post hoc power as an integral part of the trend assessment.

2.2 Adjustment of loads

2.2.1 Consider the general procedure of adjustment and trend assessment as outlined in INPUT(2) 98/5/3 and 98/5/4 with regard to its statistical implications.

2.2.2 Consider and examine the choice of the statistical model and the underlying variables for adjustment.

2.2.3 Give advice with respect to the choice of multiplicative or additive adjustment.

2.2.4 Give advice on how to measure the gain of adjustment.

2.2.5 Examine whether and when there is a risk of "over-adjustment".

2.2.6 Consider whether the use of annual adjusted loads in the trend analysis of monthly loads may become redundant.

2.3 The use of monthly data

2.3.1 Development of provisions for the use of monthly data in these trend detection methods (taking into account that any recommendations should be based on real need and best scientific judgements and should not be driven purely by statistical considerations).

In continuation of the information presented in the 1999 ACME report, Section 6.6.1 and Annex 2 contain additional material in response to this request. This material comprises information on:

1) robust smoother methods;
2) adjustment of loads;
3) analysis of monthly data;
4) use of annual mean concentration;
5) reviewing the trend-y-tector and draft guidelines.

The ACME notes that at present no single method of calculating annual adjusted loads can be recommended for all river systems/locations and all substances. However, when considering nutrients, if the input is approximately linearly related to flow/precipitation, method L1 (in Annex 2, Part 1) will perform reasonably well and the ACME recommends that it be used on a trial basis.

The ACME notes that, at this stage, it is not possible to quantify generally the gain of analysing monthly data instead of annual data for trend assessments, since this depends on the type of data and the methods applied.

The ACME also notes that the use of robust trend detection methods depends on the purpose of the assessment, as stated in Section 6.4 of the 1999 ACME report. A robust approach is not always recommended, e.g., if information about alarming recent trends is given higher priority than making a statement unaffected by incidental outliers, then a robust approach should not be used.

Finally, since the annual mean concentration does not necessarily reflect the effectiveness in reduction of inputs, not even in the long run, the ACME recommends that the use of the annual mean concentration should be avoided.

3 ADVICE AND ASSISTANCE ON EUTROPHICATION

3.1 Provision of advice and standard data products to a small group of OSPAR experts (3–5) working at the ICES Secretariat for the purposes of developing the Common Procedure for the Identification of the Eutrophication Status of the Maritime Area and contributing to further NEUT intersessional work.

The response to this request requires that the ICES Secretariat be contacted by a small group of OSPAR experts and, as indicated in Section 20.2, this contact has not yet been made. It is hoped that work on this request can be conducted during the course of 2000.

DATA HANDLING

4 To carry out data handling activities relating to:

4.1 contaminant concentrations in biota and sediments;

4.2 measurements of biological effects;

4.3 the implementation of the Nutrient Monitoring Programme.

The ICES Environmental Data Centre has handled all data submitted in 1999, covering monitoring activities in 1998. Further information on this work is contained in Section 20.1 of this report.

The ICES Oceanographic Data Centre continues to maintain as complete as possible a data set on nutrients and other oceanographic parameters in the ICES area, however, very few submissions have been received during the past year, particularly with respect to OSPAR data. More details can be found in Section 20.2.

5 To continue to establish databanks for:

5.1 phytobenthos, zoobenthos and phytoplankton species.

Section 20.3 describes progress in the development of a Biological Data Reporting Format and data entry program for the submission of data on phytobenthos, zoobenthos, and phytoplankton species under the OSPAR Nutrient Monitoring Programme. The Biological Data Reporting Format is now complete and is available on the ICES website at http://www.ices.dk/env/. The Biological Community Database is intended to be complete in the near future, and the data entry program will be ready before the end of 2000.

6 To expand the ICES environmental data reporting format to include all the reporting parameters required for each of the biological effects techniques adopted by OSPAR, where these reporting formats have not already been developed. In so doing to undertake this work in three stages:

Stage 1: the development of reporting formats for:

- *P4501A1 (EROD)*
- *PAH metabolites*
- *DNA adducts*
- *Liver histopathology*
- *Liver nodules*
- *TBT (intersex, imposex)*

Stage 2: the development of reporting formats for:

- *Metallothionein*
- *ALA-D*
- *Bioassays*
- *Fish reproductive success*

Stage 3: the development of reporting formats for:

- *TBT shell thickening*
- *Lysosomal stability*
- *Antioxidant enzymes*

Section 20.5 and Annex 10 describe the progress in the development of reporting formats for biological effects data. Further work will be conducted during 2000, but finalization of the reporting formats may need to await the final outcome of the BEQUALM Programme (see Section 7.3), which is developing quality assurance procedures for these techniques.

2.2 Scientific Peer Review of the OSPAR QSR 2000

Request

Item 1.1 of the 1999 Work Programme from the OSPAR Commission: to arrange a special ACME meeting to conduct a scientific peer review of QSR 2000 with the following aims:

a) to establish the scientific veracity of the contents;

b) to assess if the material is presented in a logical and clear sequence;

c) to determine if the methods used to back-up scientific statements are based on recognised, documented and quality controlled techniques;

d) to check if all figures and tables are correctly cited in the text and that their legend provides an adequate explanation of the content;

e) to comment on the style of illustrations, to identify if they are too complicated or lack clarity and to suggest, where necessary, ways in which they may be improved;

f) to check if all statements that should be referenced are adequately cited;

g) to identify where drafters have not complied with the agreed instructions to authors.

To present the results through the following two-stage approach:

a) to provide, following the special ACME meeting, a draft report of the review to ACG for comment;

b) to respond to ACG requests for additional information or clarification of specific points;

c) to complete, on receipt of comments from ACG, a report in a structure determined by ACG and containing the final review, and forward it for consideration at ASMO(3) November 1999.

Source of the information presented

Draft 2, version 2 of Chapters 1, 2, 3, 4, and 5, and draft 2 of Chapter 6 of the OSPAR QSR 2000, and ACME deliberations.

Status/background information

This scientific peer review of the OSPAR Quality Status Report (QSR) 2000 had originally been scheduled for late November 1999, but in June 1999 the OSPAR Commission requested ICES to postpone the review until late January 2000, owing to a delay in the preparation of the regional QSRs. This request for postponement was accepted by ICES.

According to the procedures agreed between ICES and OSPAR, the chapters of the QSR to be reviewed were to be posted on the ICES website by 17 December 1999. The texts and tables for Chapters 1, 3, 4, 5 and the first draft of Chapter 6 were received by ICES and posted by that date. Chapter 2 was received later and, owing to computer changes at ICES, was not posted until 4 January 2000. A second, completely revised draft of Chapter 6 was received by ICES on 21 January and thus was not available for review by ACME members in advance of the meeting. As the QSR 2000 is intended to be based on the five regional QSRs, the OSPAR Secretariat distributed a CD-ROM containing the complete texts, tables, and figures for these regional QSRs (rQSRs) to persons on the list of anticipated participants in the ACME meeting, as supplied by the ICES Secretariat in early December. The review of the OSPAR QSR 2000 was conducted at an extraordinary meeting of ACME held on 26 January to 2 February 2000, with the first four days devoted to this review.

The composition of experts on the ACME covered a broad range of subjects, from physical, chemical and biological oceanography, chemical contaminants and their biological effects, ecology and habitats, to fisheries biology. It was acknowledged, however, that the ACME did not have adequate expertise to cover agricultural discharges, radioactive discharges, or military activities.

The ACME reviewed each chapter as well as the entire document for balance in the coverage of topics and clarity of presentation to the anticipated readership. Each chapter was then reviewed in detail for scientific accuracy. The review in relation to the regional QSRs showed that, in certain cases, the rQSRs did not contain adequate material on which to base the overall OSPAR Convention-wide treatment of the subject; in other cases, the regional QSRs contained useful material that had not been brought forward to the QSR 2000. Where possible, when the text was considered inaccurate or incomplete, the ACME suggested corrections or new text. This was not possible in all cases, however, as there was not enough time to prepare adequate revisions. When this was the case, the ACME provided advice on how the material should be redrafted and sources of additional required information.

The overall comments by the ACME on the QSR 2000 Holistic Report are as follows:

1) According to Chapter 1, this report is intended to provide an assessment of the quality status of the maritime environment of the OSPAR area, provide an outline of the most important human activities, identify trends, evaluate the effectiveness of measures taken, and, finally, give a prioritization of human pressures. While this report covers most of the relevant ground, and appears to be comprehensive in terms of the geographical area and subject matter described, it does not meet all of these stated objectives.

2) The overall impression given by the report is that proper coordination among the chapters was not ensured. Consequently, there are numerous errors, omissions, and inconsistencies. In general, more tables and figures should be used to present information. However, many of the existing figures are not clear. The text is often too general, with few or no examples given.

3) There seems to have been over-reliance on the rQSRs as source material in Chapters 1 to 5. In some cases, information should have been sought from other published sources not dealt with in the rQSRs. At the same time, some important information in the rQSRs was not adequately used.

4) In many sections, the intended audience (laypersons, administrators, etc.) will find the text difficult to follow, while interested scientists will not find sufficiently detailed information.

5) The supporting data are uneven within and among the Regions, and often rather outdated. In turn, the information in the rQSRs has not been used uniformly in the preparation of the QSR 2000.

6) Although it would not be appropriate for every statement to be referenced, the ACME believes that it is necessary for the more important or contentious points to be backed up by key references. There are numerous occasions when important points are extracted from the rQSRs but there is no reference in the rQSR to the source of the information.

7) In many cases, it is difficult or impossible to find information on temporal trends that are important for administrators when forming opinions about the impacts of their policy decisions and for formulating new policies. There is information available on trends in the rQSRs and other readily available documents that has not been included in the various sections.

8) There is a need for a clearer Introduction to each chapter. A brief Conclusions section would also be useful so that each chapter can stand alone, if necessary.

9) In general, there are many inaccuracies in this document. In the editing stage, a thorough check on the accuracy of data, table numbers, units, and captions of tables and figures is needed. The tables are especially poor in quality and need thorough editing. The units used in the tables, figures, and text are not always consistent or in accordance with the Guidelines to Authors. The use of terminology is not consistent. Also, it should be verified that all technical terms and abbreviations are listed in the Glossary.

10) The ACME has attempted to address many of these problems, while bearing in mind that OSPAR has set a stringent limit to the overall size of the document. Where possible and necessary, improved text has therefore been suggested (and underlined), and redundant and/or incorrect material crossed out (but not deleted). Where the ACME did not have ready access to required information, it has indicated the general problem in italics, and recommended that OSPAR should obtain the material.

11) The ACME has generally not attempted to deal with editorial issues in this report unless the meaning of the text was unclear. However, it is clear that a strong editorial input will be required before the report is ready for printing.

12) Given the dependence of Chapter 6 on the final content and conclusions drawn from the other chapters, Chapter 6 will have to be carefully edited after the other chapters are finalized. The present version of Chapter 6 was incomplete, lacking the sections on overall assessment (6.3) and conclusions (6.4). This made it difficult to evaluate the overall conclusions on the environmental status which can be drawn from the QSR.

The ACME reviewed all chapters according to scientific topics, to ensure that there was consistency among the chapters in the treatment of these topics. General comments were prepared for each chapter and specific, detailed additions and amendments were inserted into the text of each chapter. Occasionally when new material was required, this was mentioned, along with the source of such material, when possible. Some movement of material between chapters was proposed, where it was felt that the material fit better in another place.

All comments were transmitted to the OSPAR Secretariat by mid-February 2000.

3 PROGRESS ON TASKS FOR THE HELSINKI COMMISSION

The present status of work on 2000 requests by the Baltic Marine Environment Protection Commission (Helsinki Commission) is given below, along with reference to the relevant sections and annexes of this report where more detailed information can be found. The requests are shown in italics and a summary of the ICES advice is then given in normal print.

CONTINUING RESPONSIBILITIES

1) *To coordinate quality assurance activities on biological and chemical measurements in the Baltic marine area and report routinely on planned and ongoing ICES intercomparison exercises, and to provide a full report on the results;*

Information on progress in the development of quality assurance procedures for biological measurements in the Baltic Sea is summarized in Section 7.1 of this report. In particular, the ICES/HELCOM Steering Group on Quality Assurance of Biological Measurements in the Baltic Sea (SGQAB) carried out work on the assessment of a new method to measure primary production and planned work to gather information on the use of the new phytobenthos monitoring guidelines. Initial discussions were held on the development of quality criteria for biological data to be used in assessments. The ACME emphasizes that there is a continuous need for the preparation of biological reference materials, the updating of taxonomic checklists, and the organization of ring tests or intercomparison exercises.

ICES expresses its deep concern over the low participation at the SGQAB meetings, and inadequate allocation of resources for other quality assurance (QA) activities carried out under SGQAB guidance. ICES stresses the importance of participation in QA activities by the Member Countries and laboratories participating in the HELCOM COMBINE Programme.

As reported in Section 7.4, additional Annexes have been completed by the ICES/HELCOM Steering Group on Quality Assurance of Chemical Measurements in the Baltic Sea (SGQAC) for the guidelines on quality assurance of chemical procedures for the COMBINE Programme. These Annexes are as follows:

- Technical Notes on Units and Conversions;
- Technical Notes on the Determination of Salinity and Temperature;
- Technical Notes on the Determination of Co-factors.

In addition, revisions and updates were prepared of existing guidelines on sampling for chemical analysis (Annex B-5 of the COMBINE Manual) and external quality assessment (Section B.6 of the COMBINE Manual). The above-mentioned material will be transmitted to HELCOM for inclusion in the COMBINE Manual.

In terms of future work, the ACME recommends that the scope of the work of SGQAC be expanded to include other parameters that are or might be included in the HELCOM monitoring programme, e.g., organic carbon and tributyltin.

2) *To evaluate every third year the populations of seals and harbour porpoise in the Baltic marine area, including the size of the populations, distribution, migration, reproductive capacity, effects of contaminants and health status, and additional mortality owing to interactions with commercial fisheries (by-catch, intentional killing);*

The review of seal and harbour porpoise populations in the Baltic Sea is contained in Section 13.1. This information shows that the marine mammal populations in the Baltic Sea are severely depleted, compared to population levels one hundred years ago. ICES is also aware that the severely depleted populations are suffering from unknown levels of by-catch mortality. Although the seal populations may be recovering, at least in part of the Baltic Sea area, the population of harbour porpoises is assumed to be at very low levels and any by-catch may pose a threat to the viability of the species in the Baltic Sea. Therefore, ICES recommends that Member Countries establish systems for obtaining estimates of total by-catches of all marine mammals in their Baltic Sea fisheries and that relevant by-catch mitigation measures be implemented where by-catches are assumed to be high.

SPECIAL ISSUES

3) *To prepare Chapter 9 on "Marine fish migratory and freshwater species in the Baltic Sea Area" for the Fourth Periodic Assessment of the State of the Baltic Marine Environment, 1994–1998, comprising:*

9.1 Exploited species

9.2 Non-commercial species

9.3 Diseases and parasites

9.4 Effects of rearing upon wild populations

9.5 Ecosystem effects of fishing activities

The draft chapter should be transmitted initially to HELCOM according to the timetable for the Fourth Periodic Assessment.

A summary of the work conducted in the preparation of this chapter is contained in Section 18.2. The draft material was reviewed and amended, including shortening the draft considerably, by ACME. Subsequently, it has been submitted to the Steering Group for the Fourth Periodic Assessment.

As the material prepared by the various contributing Working Groups needed to be shortened considerably to be able to meet the page limit allocated to this chapter, the ACME has included much of this material in its report to give a more detailed picture of the subjects covered. These subjects include fish diseases (Section 9.4 and Annex 6), the effects of mariculture activities in the Baltic Sea (Section 14.2), genetic effects of the release of cultured fish in the Baltic Sea (Section 15.1), and ecosystem effects of fishing activities in the Baltic (Section 18.1).

4) *To participate in the planning and carrying out of the proposed workshop on background/reference values for concentrations of nutrients and hazardous substances in the Baltic Sea area, in the year 2000.*

Advice on the issue of determining background/reference values is contained in Section 6.2 of this report. In particular, the ACME cautions that the terms used be clearly defined and that the means of derivation of these values be clearly explained.

5) *To provide information on the possible impact on the Baltic marine environment of dumping of fish remnants, especially regarding:*

- *size of disposal of fish remnants from fish-processing units,*
- *amounts of disposal of undersized fish by fishermen,*
- *possible secondary effects caused by dumping of fish remnants.*

Section 11 contains a compilation of estimates of annual amounts of discards and fish offal discharged into the Baltic Sea and the potential effects. While it is clear that this material is not complete, the estimates indicate that the proportion of discards in the Baltic Sea is small (3.8 % of the total fisheries excluding industrial fisheries), amounting to approximately 11 000 tonnes per year. The total amount of fish offal discharged has been estimated at approximately 15 000 tonnes per year. For both offal and discards, the largest quantities are dumped in the southwestern Baltic Sea, mainly in association with the cod fishery. Estimates of the consumption of discards and fish offal by seabirds (provided in detail in Annex 7) indicate, however, that at least 50 % of this material may be consumed by seabirds.

4 STATUS OF FISHERIES AND RELATED ENVIRONMENT OF NORTHERN SEAS

Request

A 1999 request by the Nordic Council of Ministers to prepare a report entitled “Status of Fisheries and Related Environment of Northern Seas”, summarizing existing material from reports of the Advisory Committee on Fishery Management and relevant material from ICES working groups on environmental topics.

Source of the information presented

A draft report on fisheries in northern seas prepared by P. Degnbol, based on the 1998 Report of the ICES Advisory Committee on Fishery Management; a draft report on environmental conditions in northern seas, prepared by C. Symon, based on material from ICES, HELCOM, and OSPAR including draft regional Quality Status Reports; and ACME deliberations.

Status/background information

The ACME reviewed a draft report on fisheries in the Northern Atlantic, including the areas west of Greenland, the northeast Arctic, the areas around Iceland and around the Faroe Islands, the North Sea and adjacent seas, and the Baltic Sea. Some information was also provided on deep-water and widely distributed stocks. This draft was based on material from the 1998 Report of the Advisory Committee on Fishery Management (ACFM) (ICES, 1999a). This document also contained a description of the theoretical basis for formulating fisheries management advice.

The ACME then reviewed a draft report on environmental conditions in the Northeast Atlantic including the Baltic Sea, composed of four sub-sections:

1) Oceanographic conditions and their relation to fish stocks;
2) Eutrophication and its relation to fish stocks;
3) Fish disease in relation to fish stocks;
4) Contaminants and their relation to fish stocks.

This report was prepared on the basis of material provided by ICES, including the results of a statistical analysis of fish disease data (ICES, 1999b), the HELCOM Third Periodic Assessment of the State of the Baltic Marine Environment (HELCOM, 1996), a report on the results of a temporal trend assessment of data on contaminants in the OSPAR area, and drafts of the five OSPAR regional Quality Status Reports.

The two reports were reviewed in detail by the ACME and a number of comments were made. Some material concerning oceanographic conditions in several of the regions was moved from the fisheries report to the section on oceanography, which was also revised substantially to provide background material for the fisheries information. It was agreed that much of the data on contaminants in fish and shellfish should be incorporated into figures, to make the material easier to visualize. After review of the fisheries and environmental reports, the ACME agreed to the text of these reports, noting that they would need to be merged into one overall document and edited for overall consistency in terminology and format by the ICES Secretariat. This was completed shortly after the meeting and the text was transmitted to the Nordic Council of Ministers.

References

HELCOM. 1996. Third Periodic Assessment of the State of the Marine Environment of the Baltic Sea, 1989–1993; Background document. Baltic Sea Environment Proceedings, No. 64B. Helsinki Commission, Helsinki, Finland.

ICES. 1999a. Report of the ICES Advisory Committee on Fishery Management, 1998. Parts 1 and 2. ICES Cooperative Research Report, 229.

ICES. 1999b. Report of the ICES Advisory Committee on the Marine Environment, 1998. ICES Cooperative Research Report, 233.

5 EFFECTS OF DIFFERENT TYPES OF FISHERIES ON NORTH SEA AND IRISH SEA BENTHIC ECOSYSTEMS

Request

A request from EC DG FISH, as follows: "ICES is requested to consider the report 'The effects of different types of fisheries on the North Sea and Irish Sea benthic ecosystems' (Lindeboom & de Groot, eds.) and to formulate management advice as to how the effects of the gears discussed in the report on benthic ecosystems could be measurably reduced, without unduly reducing the possibilities of catching commercially important species. ICES is invited to consider all possibilities, like establishing closed areas for bottom gears, reducing the weight of bottom gears etc.".

Source of the information presented

The 1999 report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO), and ACME deliberations.

Status/background information

In September 1998, the European Commission requested ICES to consider the report "The effects of different types of fisheries on the North Sea and Irish Sea benthic ecosystems" (Lindeboom and de Groot, 1998) and to formulate management advice as to how the effects on benthic communities of the gears discussed in the report could be measurably reduced without unduly reducing the possibilities of catching commercially important species. The advice should consider all possibilities, such as establishing closed areas for bottom gears, reducing the weight of bottom gears, or other relevant measures. The following material has been prepared in response to this request. In the material below, the report "The effects of different types of fisheries on the North Sea and Irish Sea benthic ecosystems" is referred to as the IMPACT II report.

5.1 Overall Considerations and Approach Taken

ICES acknowledges the need to pay close attention to the formal request for advice. At the same time, ICES believes that it should acknowledge the much wider interest in the question of how bottom trawling affects marine ecosystems, particularly their benthic components, and the corresponding likelihood that any advice would be interpreted more widely that just within the context of benthic taxa in the North Sea and Irish Sea. ICES has tried to be clear in all contexts with regard to the generality or restricted application of findings, conclusions, and advice, and encourages users to note when restrictions are placed on the generality of the advice which follows.

Following the content of the IMPACT II report, when the phrase "bottom trawls" is used in the advice, it refers to beam trawls and otter trawls that have been used in recent years in the southern North Sea and the Irish Sea. However, many other bottom gears are in use in various fisheries in the ICES area and beyond, and even beam trawls and otter trawls used in the North Sea and Irish Sea are classes of gears that undergo technological changes often. **Except where ICES explicitly states that conclusions apply beyond beam trawls and otter trawls, the advice should be restricted to those two gears, as they are used in the areas considered in the IMPACT II report.**

In considering the IMPACT II report, all studies and conclusions were reviewed with regard to research design, appropriateness of analytical methods, and the strength of support that the component studies provide for conclusions regarding impacts. Moreover, ICES considered a large body of other scientific literature on this topic, both from the ICES region and elsewhere. The advice differentiates whether evidence for an effect was found in the IMPACT II study areas, in other parts of the North Sea and Irish Sea, or in other seas and oceans.

To apply the review of scientific information in the IMPACT II report and additional literature directly to an evaluation of the effects of bottom trawling on benthos, ICES prepared a list of the *possible* effects that bottom trawling might have on species, communities, and habitats. It then evaluated the degree to which scientific evidence supported the occurrence of each effect first in the North Sea and Irish Sea, and then more generally. It also ranked the possible effects with regard to their seriousness and the need for corrective measures, if the effects were found to occur.

Following the same approach, ICES also listed the classes of mitigative measures which would reasonably be available to reduce the effects of fishing on benthos. The measures fell into two classes: measures which had an intrinsic spatial component and measures which were intrinsically applied at the scale of the fishery or fisheries being changed in some way. The two lists were cross-tabulated, and professional judgments were made about the degree to which each type of measure would reduce or eliminate each type of effect. From the tabulation of mitigative measures by possible effects, and the evaluation of the evidence regarding the actual occurrence of each class of possible effect in the North Sea and Irish Sea, ICES was able to identify a number of recommendations for action. Many of the recommendations require a finer scoping of specific problems before specific actions can be planned and implemented, **and the advised Priority Management Measures should not be viewed as universally applicable remedies, to be applied without further**

thought. They should be developed as well-planned mitigation programmes to address well-specified problems. Implementation of the advised **Specific Immediate Actions** could commence without undue delay, but even there programmes must be planned carefully, and with appropriate consultation.

5.2 Potential Effects of Bottom Trawls

5.2.1 Contextural considerations

This section considers the effects of bottom trawls (beam and otter trawls) on habitats and species in the shelf seas of northwestern Europe. Although effects will be specific to gear and area, ICES is confident that its classes of effects are highly applicable to these areas and elsewhere. ICES further recognizes that other types of towed bottom gears, such as dredges, can sometimes cause severe effects on the seabed. However, these gears are outside the request for advice and have been excluded from the analysis which follows.

It must be recognized 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 persons 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 they 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, dredging, land reclamation, 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.

5.2.2 Ranking the severity of potential effects

The effects of bottom trawling can be identified on 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 which occurs over a wide geographical scale will be less affected by an effect which operates only over a relatively small area. Conversely, 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, ICES has ranked the effects within these categories. The criteria used to rank the effects are based on the consideration of:

- Temporal scale: effects which cause permanent or long-lasting changes are of greater concern.
- Spatial scale: effects which occur at large spatial scales are of greater concern.
- Direction of change: declines in species or reductions in habitat features are of greater concern than increases, because of their potential irreversibility.

In several cases ICES has given two or three effects the same rank, reflecting equal concern about the effects assigned the same rank.

Other criteria for ranking could be considered (see, for example, Section 5.2.5, below), but ICES feels that these three criteria are the most important with regard to evaluating biological impacts of fisheries. ICES acknowledges the importance of social and economic effects, but is most competent to evaluate biological effects of fisheries on ecosystems. Furthermore, ICES notes that many international agreements (UNCED and FAO Codes of Conduct) consider that the pursuit of social and economic goals cannot be at the expense of conservation. ICES also acknowledges the concern that many direct effects of bottom trawls may cause additional indirect effects on other components of the ecosystem. These considerations are dealt with separately in this advice.

In developing the list of possible specific effects on benthic habitats and species (basically macrobenthos, but also fish species ecologically or spatially tied closely to the benthos), ICES also noted two general issues. These are:

- **Low-energy environments are more affected by bottom trawling**. Habitats that are frequently disturbed by natural processes (storms, tidal scour, etc.) may be little affected, or may even be unaffected, by bottom trawling. Benthic species characteristic of such high-energy habitats are generally well adapted to frequent natural disturbance. Moreover, many high-energy environments have a long history of exposure to trawling, particularly those in the southern North Sea, and species highly sensitive to bottom trawling are likely to have been depleted or locally extirpated long ago. Correspondingly, impacts of present bottom trawling on species assemblages in such high-energy environments are often likely to be small as well. Conversely, habitats and 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 communities and habitats 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.

These considerations should be kept in mind when evaluating any of the direct effects identified below in any specific situation.

5.2.3 Possible effects on habitats: description

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 important 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 I).

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, important 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 II).

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 III).

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 physical changes, which may result in habitat degradation, is reduced protection from predators of benthic species and the juveniles of commercially important fish and shellfish.

5.2.4 Possible effects on species: description

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 and, in extreme cases, to local extirpation.

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 on 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. The relative abundance of individual species can either increase or decrease depending on the relative abundances of species in the community prior to the commencement of bottom trawling.

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 example, 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 slower growth, increased likelihood of diseases, increased risk of predation, decreased competitive ability and, at a population level, reduced 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 changes in predator abundance and food supply may combine with the relative robustness that taxa with high turnover rates often have 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.

5.2.5 Potential effects on food-web and ecosystem properties

Fishing has had two major direct effects on the food web structure of the North Sea and Irish Sea. First, it has added a further highly efficient and effective "predator"—the fisheries—to the apex of the food web. Second, by applying fishing mortality differentially among species, fishing has altered the relative abundance of the species in the rest of the food web. However, the magnitudes and seriousness of the consequences of these direct effects on food web and ecosystem properties are highly uncertain, and widely debated.

There is a great deal of theoretical argument about what the consequences *might* be. The changes in relative abundance of species could bring about changes in predator-prey interaction strengths, and so alter the pathway of energy flow through the food web in two different ways. First, there are changes 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, from one species of piscivorous demersal roundfish to another. The second class of possible food web alterations involves changes where the energy flow through the food web takes entirely different pathways, involving groups of species performing quite different functional roles, for example, a shift from gadoids to a pelagic-dominated system. In all the possible examples of such major food web changes reviewed by ICES, it was not fishing *per se* that caused species replacements or changes in functional group composition. Rather it was *over*-fishing, and usually accompanied by a major environmental event, making it impossible to determine the role of over-fishing alone in causing the change.

The above discussion addresses "top-down" effects: the effects of changes in predator populations on the populations of their prey. However, bottom-up food web effects may also have occurred. Beam and otter trawling disturbs the seabed, causing nutrients locked up in the sediments to be released into the water column. This could 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. The consequences of such changes in the food web are uncertain and situation-specific, and different analytical or modelling approaches may make different predictions about even the direction of the effect on particular species higher up in the food web.

5.3 Evidence for the Various Classes of Effects

5.3.1 General conclusions regarding strength of evidence

Following the organization of the IMPACT II report, ICES reviewed the evidence for each class of direct effects separately for 1) studies of gears, fleets, and physical impacts of gears; 2) studies of direct mortality on benthic organisms; 3) studies contrasting fished and unfished areas; and 4) long-term studies. The evaluations from each type of evidence are combined in the advice, however, and summarized in Table 5.3.1.1. The table highlights both the overall judgement by ICES of the strength of the evidence for each effect, and whether evidence is specifically available for the North Sea and Irish Sea, the areas specified in the request for advice. To

Table 5.3.1.1. Summary of the information on evidence for the various possible effects of bottom trawling on species (macrobenthos and fish closely associated ecologically or spatially with benthos) and habitats. Cell entries reflect ICES decisions on the weight of evidence. For many reasons related to study design, implementation, and analysis, or to true differences among specific situations, individual studies may differ in their conclusions regarding various effects of bottom trawling.

Type of effect	**Strength of evidence**		**Type of evidence**		**Duration of effect**	**Environments affected**	
	North Sea and Irish Sea	**Other areas**	**Long-term studies**	**Experimental studies**		**High energy**	**Low energy**
Effects on habitats							
Removal of major physical features (Habitat Priority I)	Weak–moderate	Strong	XXX		Permanent	XXX	XXX
Reduction of structural biota (Habitat Priority I)	Weak–moderate	Strong	XXX	XXX	Years to decades	X	XXX
Reduction in habitat complexity (Habitat Priority II)	Weak	Weak		XXX	Days to several months	Negligible	XXX
Changes sea floor structure (Habitat Priority III)	Strong	Strong		XXX	Days to several months	Negligible	XXX
Effects on species							
Reduction in geographic range (Species Priority I)	Moderate*	Strong*	XXX		Years to decades	XXX	XXX
Decrease in species with low turnover rates (Species Priority I)	Weak–moderate	Moderate –strong	XXX	XXX	Years to decades	X	XXX
Fragmentation of populations (Species Priority I)	None	Weak	XXX		Years to decades	XXX	XXX
Changes in relative abundance of species (Species Priority II)	Strong*	Strong*	XXX	XXX	Days to many years	XXX	XXX
Fragile species more affected (Species Priority III)	Weak	Weak		XXX	Unclear	X	XXX
Surface-living species more affected than burrowing species (Species Priority III)	Weak	Weak		XXX	Weeks to few years	Unclear	Unclear
Sub-lethal effects on individuals (Species Priority IV)	Moderate–strong	Moderate –strong		XXX	Weeks to few years	X	XXX
Increase in species with high turnover rates (Species Priority V)	Moderate	Moderate	XXX		Months to few years	X	XXX
Increase scavenger populations (Species Priority V)	Weak	Moderate	XXX	X	Days to months	XXX	XXX

X = Effect can be present, but is rarely large. XXX = Effect is often present, and can be large. An empty cell = No evidence from the class of studies was found for the effect. Unclear = Few studies provide information about whether the type of environment affects the likelihood or severity of the effect. * = Evidence of **change** in populations and ranges is moderate or strong. Because environmental conditions have changed over the period of fishing, and many stocks are exploited by several fisheries, it is usually difficult to unambiguously partition the contribution of a single factor, such as bottom trawling, to the quantified change.

give perspective on space and time scales of the possible effects, the expected duration of each type of effect is also indicated, as is the likelihood that the effect, if present, would be expected mainly in low-energy environments or in both low-energy and high-energy environments. **ICES stresses that these designations are generalizations of scientifically complex issues using the results of diverse and often imperfect studies. Exceptions to the generalizations are likely to occur, and specific cases should be examined before it is assumed that a particular effect will be found in a particular area, habitat, or species.**

Notwithstanding the qualifications above, **ICES concludes that there is evidence for the occurrence of all the effects** listed in Section 5.2, and **the evidence is strong for the two higher priority effects on habitats,**

and all the higher priority effects on species except the fragmentation of populations. However, it should be noted that in some cases the strong evidence comes from studies outside the North Sea and Irish Sea, and several of the effects are of small or negligible concern in high-energy environments. Given evidence for the effects of higher priority, ICES concludes that appropriate mitigative actions are warranted. More information on the sources and types of evidence is presented below, and developed in more detail in ICES (2000). Considerations in developing appropriate mitigative actions are described in Sections 5.4 and 5.5, below.

5.3.2 Summary of evidence for each type of effect

5.3.2.1 Effects on habitats

Bottom trawling can remove some physical features.

There is no evidence for this effect in the IMPACT II report, or in recent studies of the southern North Sea and Irish Sea. However, other scientific reports in other areas, including Norway and New England, have demonstrated that physical features, such as cobbles and boulders, are moved by trawling. One can infer that it is likely that bottom trawls have removed some physical features in the North Sea and Irish Sea in the past, and there are numerous anecdotal reports that this has been the case.

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

There is no evidence for this effect in the IMPACT II studies, which were principally carried out in soft sedimentary environments where such biogenic features are not found. In other areas, including Australia, Norway, and New England, effects of trawling have been demonstrated by comparative experiments in habitats characterized by biogenic features such as sponges, eelgrass, worm tubes, bryozoa and hydrozoa. For example, the destruction of biogenic structures such as tubes and burrows and the loss of flocculated organic matter on the sediment surface was noted 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. Long-term studies have provided strong evidence for the effect from elsewhere, particularly New England and Alaska, 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; such organisms include, e.g., hydroids, bryozoans, sponges, and serpulid worms. 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.

The findings presented in the IMPACT II report support this effect of bottom trawling, which has also been found in experimental studies in other areas, including the Northwest Atlantic, North Pacific, and Australian waters.

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

The findings presented in the IMPACT II report support this effect of bottom trawling. In the IMPACT II studies, sediment characteristics in trawled plots in both the Loch Gareloch and West Gamma experiments may have changed as a result of trawling. However, problems with the experimental design make it difficult to definitely attribute changes in sediment characteristics to trawling impacts. Subsequent experimental studies in the North Sea and on the Grand Bank have demonstrated that beam trawling can make sediments more homogeneous. However, there is support elsewhere for effects of bottom trawling on the physical structure of habitats in the opposite direction as well, at least in the short term. Off Canada, otter trawl doors have been found to create berms and furrows which 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.

5.3.2.2 Effects on species

Bottom trawling can cause a reduction in the geographical range of species.

There is some evidence from the long-term studies in the IMPACT II report to support this conclusion. The 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 studies from the literature have suggested these range effects for the North Sea, the Northwest Atlantic, and many other areas, but the possible contribution of other environmental and/or anthropogenic factors to the changes in distribution cannot easily be discounted. In general, experimental studies are unlikely to be informative about the role of bottom trawling in reduction of the geographical range of a species. Experimental studies designed to compare fished and unfished areas usually will have the two types of areas sufficiently close that they will provide little information about changes in the range of a species.

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

In all three experimental studies reported in the IMPACT II report, the majority of species had high population turnovers. For those species with a low population turnover, the data provided no evidence that they had been particularly severely affected by trawling activities during the studies. However, another later study in the North Sea did find a decrease in species with low turnover. For long-term studies, results consistent with this effect have been reported for the North Sea, the Northwest Atlantic, and other areas. Again, usually trends in populations are documented, but the relative contributions of fishing and other factors to causing the trends are highly uncertain.

Bottom trawling is patchy and can cause fragmentation of populations.

Little literature was found supporting the occurrence of this effect for the North Sea and Irish Sea or for other areas, but few long-term studies focused directly on investigating potential population fragmentation. In general, experimental studies will provide little information about fragmentation of populations. Experimental studies designed to compare fished and unfished areas will have the two types of areas sufficiently close, and will have monitoring restricted to selected treated and untreated areas, so they will be unlikely to detect larger scale changes in patterns of distribution of species.

The relative abundance of species is altered by bottom trawling.

There is limited and indirect experimental evidence (mechanisms which could lead to this result) to support this conclusion in the IMPACT II report, however no analyses directly investigating this question were performed with the data. Direct mortality studies generally would not report information on this type of impact, although changes in relative abundance could be inferred from studies reporting differential mortality due to bottom trawls. In the Loch Gareloch experimental study, differences were found between treatment and control plots. However, although this experiment showed some effects on abundances of benthos, because the experiment was pseudo-replicated, these changes cannot be unambiguously attributed to trawling. Other studies in the North Sea and elsewhere, including the Northwest Atlantic and Alaska, have found changes in abundance (generally decrease) of infauna.

There is clear evidence from long-term studies reported in the IMPACT II report that species relative abundance has changed over a long period. The IMPACT II report noted correctly the problems with concluding that bottom trawling is the only cause of this variability, but could not evaluate further the possible roles of alternative factors. Subsequent studies of North Sea benthos found clear patterns of change in the macrofaunal communities in three areas between the early 1920s and the late 1980s. The lack of change in two other areas was interpreted as evidence that the observed patterns were not part of a broad-scale environmental change. 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 only 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 of which are indirect effects of fishing, may be at least as important as direct effects.

Long-term variations in fish species composition, leading to change in species diversity, have been demonstrated in the northwestern North Sea, the Barents Sea, the Northwest Atlantic, the Gulf of Thailand, and many other areas. Most fish communities that have been studied for a decade or more show changes in the relative abundance of the species present. The degree to which these changes can be attributed to the effect of fishing is unclear in every case.

Fragile species are more affected by bottom trawling than robust species.

ICES found no conclusive evidence to support this statement in the IMPACT II report. In the Loch Gareloch study, it was stated that several fragile species such as *Metridium senile* declined in abundance due to trawling. However, this conclusion can be questioned. In the Loch Gareloch study, the authors acknowledge that there are few species, such as corals and sponges, which can be regarded as fragile. It is not clear whether *Metridium senile* is as vulnerable to trawling disturbance as claimed in the IMPACT II report. In the wreck studies reported in the IMPACT II report, none of the species recorded there can be regarded as fragile. In other work in the Northwest Atlantic, the vulnerability of bivalve molluscs to damage could be ranked according to mechanical shell strength and it has been demonstrated that thin-shelled bivalves were more easily damaged by otter trawl doors than thick-shelled bivalves. However, certain size classes within species of molluscs have been shown to be protected from some bottom gears 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. A description of the classifications and qualifiers as to size would be necessary. However, for upright brittle species such as hard corals, this statement would generally apply.

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

This is supported by studies of direct mortality reported in the IMPACT II report, and other literature for the North Sea, Irish Sea, and elsewhere. This effect is a direct consequence of the penetration of the gear. In the Loch Gareloch and wreck studies, the majority of the fauna were 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 the IMPACT II report provide little evidence that this is the case. Other studies in the North Sea have measured higher mortality rates for surface-dwelling benthos than for deep-burrowing fauna.

Bottom trawling can have sub-lethal effects on individuals.

There is only limited evidence for this in the IMPACT II report. Other work in the North Sea and Irish Sea has shown that the proportion of two species of starfish with damaged or regenerating limbs increased with increasing fishing intensity. Various degrees of damage on epibenthic and shallow infaunal bivalves have been reported for different types of mobile fishing gear in other studies in the North Sea, and off Alaska and Australia. Trawling has been observed to damage the shells of the bivalve *Arctica islandica* in the North Sea. 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, it was possible to correlate increased shell damage with increased beam trawling activity between 1972 and 1991.

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

In the experimental Loch Gareloch study of the IMPACT II report, the authors concluded that several species of polychaetes became very abundant in treatment plots following trawling. ICES reviewed the evidence and considers that the conclusion is not justified from the information presented. The literature review did not locate any sound experimental studies which have found increases in populations identified *a priori* as having high rates of turnover within disturbed areas.

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 long-term studies in the German Bight. 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 these conclusions. There is evidence from studies of demersal fish populations in many areas of the world that also suggests that populations of small-bodied fish have increased over a long time period. Where data on fishing effort are available, these trends correlated with changes in fishing effort. Although these observations have been noted for fish populations over the long term, ICES did not find data which would allow this effect to be demonstrated in the benthos.

Bottom trawling favours populations of scavenging species.

Although the IMPACT II report does not provide conclusive evidence of this phenomenon, other studies support this conclusion for the North Sea and elsewhere. In the West Gamma experimental study, a greater number of scavengers was found in the near vicinity of the wreck, but it is not possible to attribute these differences to trawling. However, experimental studies in the North Sea and elsewhere have documented that invertebrate and fish scavengers enter areas which have recently been trawled. ICES found no other relevant data for benthos from the literature on long-term studies. The effect is, however, well documented for other parts of the North Sea ecosystem, including meroplankton and scavenging seabirds (ICES, 1996).

5.3.2.3 Other potential causative factors

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; Hollingworth, 2000), as well as the information reviewed in the IMPACT II report. There are also a number of other analyses of demersal fish and benthic invertebrate data sets from the North Sea, other ICES waters, and other parts of the world, which are relevant to the issues discussed here.

Changes in fishing practices are far from being the only changes to have occurred in the North Sea and Irish Sea during the last few decades. Variations in sea surface temperature and salinity, the strength of Atlantic inflow in the north, and dissolved inorganic phosphate concentrations are just some of the physical and chemical 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 variations have 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 and Irish Sea has increased over the course of the Twentieth Century, the development of the fishing industry there 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.,* 1999; 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 5.3.2.3.1. 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 partitioning these relationships would be complex.

Table 5.3.2.3.1. 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*. (1996)	Decrease
Three bearded rockling (*Gaidropsarus vulgaris*)	Rijnsdorp *et al*. (1996)	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 (*Tricopterus luscus*)	Heessen and Daan (1996)	Increase
	Corten and van de Kamp (1996)	No trend
Poor cod (*Trisopterus minutus*)	Rijnsdorp *et al*. (1996)	Decrease
	Heessen and Daan (1996)	Increase
	Greenstreet and Hall (1996)	No trend
	Corten and van de Kamp (1996)	Increase
Hake (*Merluccius merluccius*)	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*. (1996)	Decrease
	Heessen (1996)	Possible increase
Grey gurnard (*Eutrigla gurnardus*)	Rijnsdorp *et al*. (1996)	Decrease
	Heessen and Daan (1996)	Increase
	Greenstreet and Hall (1996)	Decrease
Tub gurnard (*Trigla lucerna*)	Rijnsdorp *et al*. (1996)	Possible decrease
	Heessen (1996)	No trend
	Corten and van de Kamp (1996)	Not determinable
Bullrout (*Myoxocephalus scorpius*)	Rijnsdorp *et al*. (1996)	Possible decrease
	Heessen and Daan (1996)	Increase
Hooknose (*Agonus cataphractus*)	Rijnsdorp *et al*. (1996)	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*. (1996)	Increase
	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 gunnellus*)	Rogers and Millner (1996)	Decrease
Catfish *(Anarhichas lupus*)	Heessen (1996)	Increase
Lesser weever (*Echiichthys vipera*)	Rijnsdorp *et al*. (1996)	Decrease
	Heessen (1996)	Increase
	Rogers and Millner (1996)	No trend
Greater weever (*Trachinus draco*)	Rijnsdorp *et al*. (1996)	Decrease

Table 5.3.2.3.1. Continued.

Species	Study	Trend in abundance
Dragonet (*Callionymus lyra*)	Rijnsdorp *et al.* (1996)	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.* (1996)	Decrease
	Heessen (1996)	Possible increase
Witch (*Glyptocephalus cynoglossus*)	Heessen (1996)	No trend
	Greenstreet *et al.* (1999)	Decrease[2]
Long rough dab (*Hippoglossoides platessoides*)	Rijnsdorp *et al.* (1996)	Decrease
	Heessen and Daan (1996)	Increase
	Greenstreet and Hall (1996)	No trend
Halibut (*Hippoglossus hippoglossus*)	Heessen (1996)	Possible increase
Common dab (*Limanda limanda*)	Rijnsdorp *et al.* (1996)	Decrease
	Heessen and Daan (1996)	Possible increase
	Greenstreet and Hall (1996)	No trend
Lemon sole (*Microstomus kitt*)[1]	Rijnsdorp *et al.* (1996)	Possible decrease
	Heessen and Daan (1996)	Increase
	Greenstreet and Hall (1996)	Increase
Solenette (*Buglossidium luteum*)	Rijnsdorp *et al.* (1996)	Decrease
	Rogers and Millner (1996)	No trend
	Heessen (1996)	Not determinable

1 This study dealt exclusively with the North Sea where lemon sole is not targeted specifically, although they are usually landed if taken in the by-catch.
2 A decrease is suggested in all four of the regions studied.

Table 5.3.2.3.1 (supplement). Type of study, spatial and temporal coverage, and gears used in the studies cited in Table 5.3.2.3.1.

Study	Study type	Period(s)	Gears	Spatial coverage
Rijnsdorp *et al.* (1996)	Two-period comparison	1906–1909, 1990–1995	OT20[1] vs. 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]	Southeastern 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 vs. AOT[5]	Northwestern North Sea
Greenstreet *et al.* (1999)	Time series	1925–1996	OT48 vs. 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.

5.3.3 Evidence for effects on food-web and ecosystem properties

ICES has not reviewed and summarized the evidence regarding effects of fishing on food webs and ecosystem properties in this advice, nor did the IMPACT II report. Rather, as noted above, ICES has concluded that any effects of fishing on food-web and ecosystem properties would be indirect ones; consequences of the direct effects are reviewed above. Compared to the direct effects, there is much less scientific consensus on both the theory predicting the occurrence of these indirect effects, and what would constitute empirical evidence from field studies for or against the presence of these effects. This should not be considered a constraint on developing conservation programmes by managers, however. Because such effects, if they occur, are consequences of the direct effects on species and habitats, measures to reduce the direct effects will also move these effects in the direction of better conservation and sustainability. The increased risk to conservation posed by effects on food-web and ecosystem properties simply adds greater motivation for timely implementation of effective measures to reduce the direct effects.

5.4 Mitigating the Effects of Bottom Trawls

There are a number of common sense issues that 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 (e.g., physical features of the sea floor in high-energy environments) to centuries (e.g., biogenic structural features in low-energy environments). 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 generally monotonic relationship between the intensity of trawling and the degree of change in the benthic environment. Change from pre-fishery conditions is expected to increase rapidly with initial increases in effort, and at some point further increases in effort would be expected to cause progressively less and less incremental change. However, the shape of the curves will be highly dependent on specific conditions, including the history of natural disturbance, the type of gear used, and the habitats, species, or assemblages of concern.
- 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.
- The adoption of measures to mitigate impacts may be encouraged by adding economic incentives for their implementation.
- **All the recommended management measures and actions listed below are to be viewed as general advice, and have not been tailored to any specific situation. Recommendations should not be viewed as universally applicable remedies, to be applied without further thought. They should be developed as well-planned mitigation programmes to address well-specified problems.**

All of these issues should be kept in mind when considering programmes to use any of the measures below to reduce effects of bottom trawling (or other types of fishing) on the benthos.

At the end of the explanation of each class of mitigative measure is the consensus view developed by ICES with regard to the effectiveness of the measure at reducing effects of bottom trawling on benthic habitats and species. Because each class of measure can be implemented in many different ways, the generic statement on effectiveness will always have exceptions, often in either direction. Programmes to implement measures with excellent potential can be designed or implemented badly, and produce no benefits. Likewise, some measures without promise of significant benefits globally might be quite effective under very specialized circumstances. Nonetheless, Table 5.4.1, which summarizes the information in the following text, should be an informative guide when viewed as general guidance with regard to which types of measures should be considered more appropriate for reducing particular types of effects.

5.4.1 Types of measures to reduce effects

5.4.1.1 Effort reduction

The single most effective measure to protect marine ecosystems is to significantly and permanently reduce fishing effort. For some specific effects of fishing, some of the technical measures described below will be highly effective. Likewise for some particular areas, such as some high-energy habitats, reductions in fishing effort may result in few conservation benefits to species already adapted to frequent disturbance. 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 other types of mitigation measures, even if implemented effectively, might not be permanent. 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.

5.4.1.2 Gear substitution

Replacing bottom trawls (beam and otter trawls) with static gears (gillnets, tangle nets, longlines, pots, traps, etc.) may reduce the degradation of sensitive habitats and may provide full and effective protection for habitats. In some cases, gear substitution can also reduce mortality of non-target species because static gears usually are more selective and generate a smaller proportion of unwanted catch than mobile gears. 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.

5.4.1.3 Change in 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 rapid on-board sorting and return of by-catch to the sea) may in principle reduce the impact of the gears. It is recognized that, because of difficulties in enforcement, it is particularly important to obtain the agreement and understanding of fishers to any required changes in gear usage. These measures are

Table 5.4.1. Summary of the expected 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 that are not inherently spatial, the degree of mitigation is related to the extent of implementation. In assessing their effectiveness, ICES has judged each measure on its potential to provide protection against the effects listed and identified in Section 5.2. For mitigation measures with an inherent spatial dimension, ICES has 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 the extent of implementation						**Mitigation with inherent spatial dimension**			
Effects (See Section 5.2)	**Reduce effort[1]**	**Gear substitution[2]**	**Gear usage[3]**	**Gear modifi-cation (light/ novel)[4]**	**Gear modifi-cation (select)[5]**	**By-catch quota[6]**	**Spatial closure**	**Real time closures**	**Improve habitat**	**Species adjust-ment**
HABITATS										
Physical* (H-I)	–	C	–	–	–	–	C	–	C	–
Biogenic (H-I)	–	C	–	–	–	–	C	–	E	M
Complex (H-II)	E	C	–	M	–	–	C	–	M	–
Structure (H-III)	E	C	–	M	–	–	C	–	–	–
SPECIES										
Range (S-I)	E	E	M	M	M	M	M	–	M	M
Low turnover (S-I)	E	E	M	M	M	M	M	–	M	M
Fragment (S-I)	M	E	M	M	M	M	–	–	M	–
Relative (S-II)	M/E	E	–	E	E	–	M/E	M	–	–
Fragile (S-III)	E	C	–	M/E	–	M	M/E	M	M	–
Surface (S-III)	E	C	M	M/E	–	–	M/E	M	–	–
Sub-lethal (S-IV)	E	C	M	M/E	E	–	M/E	M	–	–
Small spp. (S-V)	M/E	E	–	M/E	M	–	E	–	–	–
Scavenger (S-V)	E	C	M	M/E	E	E	M	M	–	–

KEY: – = none; M = marginal protection; E = effective protection; C = complete protection; * = no natural recovery potential.

1 Assuming a 50 % reduction in effort; 50 % is used for illustration and is not to be considered a specifically recommended percentage 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 that 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 sea floor.
5 Assuming modifications such as excluder/escape devices which increase selectivity, and the survival of species.
6 Assuming that quotas are set at an appropriate level to provide protection of vulnerable populations.

ineffective in reducing impacts on habitats, and are only marginally effective in reducing impacts on populations.

5.4.1.4 Gear modifications

Lighten the gear

Increasing the weight of gear components which stimulate target organisms from the seabed (e.g., tickler chains) is used as a means to improve the efficiency of some gears. 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.

Use novel techniques

Modifications to bottom trawls can incorporate new technologies or designs to reduce the effect of gear components which contact the seabed. These mitigation measures are likely to be effective in offering marginal or effective protection for small benthic species and scavengers, but may be largely ineffective on many populations of fish. Any development of novel gear should not be allowed to further impact the marine environment or spread such effects to previously inaccessible areas.

Develop more selective gears

Size selectivity of bottom trawls can be enhanced by a wide range of alterations to gear design. Most fisheries are multispecies 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 will have positive effects on populations, but will be largely ineffective in reducing the impact of gears on habitats.

Any gear modifications that encourage fishers to change their use of the gear to ways which dissipate the benefits, will ultimately not be of value.

5.4.1.5 Spatial closures

The complete closure of parts of the sea to exclude beam trawls and otter trawls 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.

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 closure needs to be evaluated on a case-by-case basis, and difficulties may be partly resolved by area zonations (that is, partitioning an area and segregating gears into different portions).

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, areas which are permanently closed for a full twelve months (i.e., not seasonally) are likely to be most effective. Despite the 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.

5.4.1.6 Real time closures

The temporary closure to bottom trawls of an area in 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. 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 realize benefits.

5.4.1.7 Improvement of 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 5.4.1). 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.

5.4.1.8 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 the 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 benthic biota, other than a few commercial shellfish species.

It has been suggested that the imbalance to marine ecosystems caused by human exploitation can be artificially readjusted 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, have 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 of very limited effect.

5.4.1.9 By-catch quotas

This is similar to a catch quota in that it sets a limit on non-targeted marine organisms (either fish or 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.

5.5 Achieving Reductions in Effects

5.5.1 Context

ICES lists several changes which can contribute to meaningful reductions in the effects of bottom trawls on the benthos of the North Sea and Irish Sea (and, more generally, contribute to the reduction of ecosystem effects of fishing). The series of measures below, if implemented successfully, would reduce the effects which ICES has 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 Fisheries Policy of the EU. However, ICES notes that a number of changes to the governance system of marine ecosystems in the North Sea are also being discussed (IMM, 1997). ICES believes that many of these changes in governance would greatly facilitate the achievement of major reductions in the effects of fishing on marine ecosystems.

None of the priority management measures are fully developed, with specific areas, species, fleets or magnitudes of change specified. Also, as illustrated in Table 5.3.1.1 and Section 5.2, the ecosystem effects caused by bottom trawling differ greatly among habitats, species, and types of fishing gears, and general recommendations are not global remedies. To address area issues with specificity would require detailed data. More importantly, however, to prepare recommendations to enhance the conservation of specific species, habitats, or sites, or to change harvesting allocations 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. However, until such priorities are set, specific advice can only be provided in response to specific questions. Given a specific question and necessary specific data on the habitat, area, or species, addressed in the question, the advice below should provide the general framework for preparing detailed advice on the types and magnitudes of management actions to be taken.

5.5.2 Priority management measures

The order of presentation reflects the priority given to each measure by ICES. It is stressed that the conservation benefits of any measure will depend on the quality of its design and implementation.

5.5.2.1 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, nor would a single percentage reduction be appropriate for all situations. However, for stocks with good analytical assessments, ICES generally considers effort reductions of at least 20 % to be required, in order for the consequences of effort changes to be measurable in the target species. The population status of benthic species is usually more poorly quantified than the status of target species. Correspondingly, ICES expects that reductions of at least 30 %, and often much more, relative to effort in the preceding few years, would be necessary in order to result in measurable benefits.

The benefits of major reductions in effort could include greatly mitigating the effect of reducing the range of species, and partially mitigating 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, as well as reducing sub-lethal 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 the full benefits of other measures proposed below to reduce effects on habitats, and if effort reduction led to the fleet never visiting some areas exploited at present, there could be habitat benefits in the medium term.

At fishing effort levels of the late 1990s, many target species were outside or close to Safe Biological Limits, with exploitation rates often close to or in excess of F_{pa}. Therefore, effort could be reduced greatly without the fisheries suffering major reductions in catches in the medium term. Moreover, the direct benefits to the target stocks would often result in increased catches in future years. ICES 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.

5.5.2.2 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 (see ICES (1994) for advice regarding the complexities of properly siting closed areas). The size of the closed area will depend on the scale of the feature which is to benefit from the closure. The nature of the closure (overall or gear-specific) will also depend on the conservation objectives that the closure is intended to achieve. There may be the 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 ecological value and vulnerability. Closures of areas used by fisheries are expected to displace that effort into other areas. Programmes to ensure that the displaced effort is either removed from the fishery, or relocates 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, and surface-living species, and sub-lethal effects on 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 can functionally exclude bottom gears from parts of their range of operation. In this way, their benefits are very similar to those of closed areas for habitats and sedentary species, unless the zoning system allows the operation of other gears with ecosystem impacts similar to those of bottom trawls.

Closed areas do not reduce the overall opportunity to harvest commercially important fish, unless a substantial portion of the range of the target stock is closed. However, they often concentrate fishing effort relative to its geographical distribution 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.

5.5.2.3 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 the by-catch of these species is low in the alternate gear, benefits can be large. The relative abundance of species will still be changed by fishing with the alternate gear which has been introduced, 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 in or elimination of 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 that can have large economic or social impacts.

5.5.2.4 Gear modifications

There are cases in which gear modifications have been very effective in reducing direct mortality on specific by-catch species, for example, sorting grids in *Pandalus* trawls. Moreover, gear modifications that 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. To be accepted by industry, gear modifications should maintain a high catchability of the target species while reducing mortality inflicted on by-catch species. Unless impact of the gear on 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.

5.5.2.5 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.

5.5.2.6 Governance changes

In its review, ICES noted that the current governance system of marine ecosystems in the ICES area may present challenges to rapid progress on some of the measures advised to reduce effects of bottom trawls on benthic systems. ICES calls special attention to the conclusions from the 1997 Intermediate Ministerial Meeting on the Integration of Fisheries and Environmental Issues, which outlined a broader conceptual basis for governance in the North Sea (IMM, 1997). ICES believes that progress on the full framework for changes to ocean governance would facilitate implementation of the measures in Sections 5.5.2.1 to 5.5.2.5, above. In particular, ICES believes that progress under the IMM 97 Statement of Conclusions Guiding Principles 2.6 ("further integration of fisheries and environmental protection, conservation, and management measures, drawing on the development and application of an ecosystem approach …"), 2.7 ("integration of environmental objectives into fisheries policy"), and 2.9 ("involvement of fishermen and other relevant parties in the decision-making process") will contribute to, and may be necessary for, achievement of significant reductions in the effects of bottom trawling on benthic systems.

5.5.3 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, in light of the evidence for these effects, as summarized in Section 5.3, above, the Precautionary Approach requires that immediate action be taken, 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 actions to promote conservation, progress can be made on improving our understanding of the threats to the habitats or species, and refining the measures used to ensure their conservation.

5.5.3.1 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 the use of other types of gears was also restricted. Implementation would be facilitated by consultation with the fishing industry.

5.5.3.2 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 of fishing 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.

5.5.3.3 Strengthen interactions with groups working on conservation of these ecosystems

Competent management agencies should expand their cooperation with the spectrum of initiatives now under way to identify and protect species and habitats at risk in the North Sea and Irish Sea (and elsewhere). As these initiatives identify specific species or habitats which require conservation action, management agencies should implement effective conservation measures swiftly. The type of initiatives of particular interest in the ICES area include, but are not restricted to, components of Natura 2000 and the OSPAR Annex V Strategy, 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.**

5.5.3.4 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.

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6 MONITORING TECHNIQUES AND GUIDELINES

6.1 Biological Effects Monitoring

6.1.1 Influence of fluctuations in salinity on biomarker and bioassay responses of marine organisms

Request

This is part of the on-going work of ICES to improve the tools available for monitoring the biological effects of marine contaminants.

Source of the information presented

The 2000 report of the Working Group on Biological Effects of Contaminants (WGBEC) and ACME deliberations.

Status/background information

Marine monitoring programmes often involve organisms from environments such as estuaries and intertidal zones where salinity can fluctuate. Biomarkers are used in these biomonitoring programmes to understand the environmental impact of anthropogenic contaminants on marine organisms, but only a few studies have investigated the influence of fluctuating salinity conditions on the responses of biomarkers in marine vertebrates and invertebrates. However, these studies show that biomarker-salinity interactions can undoubtedly occur. It is therefore possible that fluctuating salinity may cause poor agreement between the results of different monitoring programmes, and may confound results obtained within individual programmes.

An overview of the literature (Table 6.1.1.1) lists approximately 25 articles dealing with the effects of contaminants under conditions of fluctuating salinity. However, many of these studies did not account for the effects or influence of salinity on the bioavailability of the contaminants in question (e.g., metal speciation), making the data difficult to interpret. In general, few studies have examined the effects of fluctuating salinity, despite its known influence on the activity of osmoregulatory systems (e.g., enzymes such as Na/K ATPase) and several hormones (e.g., prolactin and cortisol, depending on the organism and the severity of the changes in salinity). There are also indications that changes in salinity (and temperature) are cues for reproductive cycles in bivalves, and they may play a similar role in other estuarine organisms. Finally, the effects of fluctuating salinity have not been studied for all biomarkers. For example, there do not appear to be any studies that have examined the effects of salinity on the induction of vitellogenin in male fish.

Need for further research or additional data

More experimental work is required to model the effects of fluctuating salinity on the susceptibility of estuarine and brackish water organisms to contaminants.

Recommendations

It is clear from the discussion above that the effects of fluctuating salinity (and other abiotic and biotic factors) on the status of biomarkers should form part of validation studies and quality assurance/quality control (QA/QC) programmes for these marine environmental assessment tools.

ICES therefore recommends that validation programmes should be conducted for biomarkers (and bioassays) to account for the effect of interfering factors. The influence not only of salinity but also of temperature, abiotic stress, seasonal variations, spawning, and oxygen levels on the response of biomarkers and bioassays must be better understood.

6.1.2 Use of *in situ* bioassays for evaluating effects of contaminants

Request

This is part of continuing ICES work to improve the tools available for monitoring biological effects of contaminants in the marine environment.

Source of the information presented

The 2000 report of the Working Group on Biological Effects of Contaminants (WGBEC) and ACME deliberations.

Status/background information

In situ bioassays have been in use since at least the 1880s when Albert Günther measured mortality in fish and shrimps held in cages in the Thames estuary in order to discover whether sewage and gasworks discharges were damaging fisheries (Wheeler, 1979). Such bioassays may be defined as deployment of a biological "system" at the site of choice in the field environment. The biological system may be a small community of organisms, individuals, or in some instances colonizing substrates, and the endpoints now available are numerous, e.g., biomarkers, physiology, behaviour, growth, reproduction, and mortality. *In situ* assays have an important role in bridging the gap between laboratory experimentation and field validation. They can offer a

Table 6.1.1.1. Literature on the effects of salinity fluctuations on biomarkers.

No.	Biomarker	Contaminant	Authors	Year
1	HSP 60	Crude oil, oil dispersant, naphthalene	Wolfe, M.F., Olsen, H.E., Gasuad, K.A., Tjeerdema, R.S., and Sowby, M.L.	1999
2	HSP 70	Cd, organic compounds	Werner, I., and Hinton, D.E.	1999
3	MT	Cd, Cu, Zn,	Mouneyrac, C., Amiard, J.C., and Amiard-Triquet, C.	1998
4	MT	Cu, Cd	Amiard-Triquet, C., Rainglet, F., Larroux, C., Regoli, F., and Hummel, H.	1998
5	Lysosomal membrane stability	Cu	Ringwood, A.H., Conners, D.E., and Hoguet, J.	1998
6	Parasites	Metals and organic contaminants	Landsberg, J.H., Blakesley, B.A., Reese, R.O., McRae, G., and Forstchen, P.R.	1998
7	Catalase, GST, MDA, AChE	Cd, Cu, Zn, aromatic compounds	Amiard-Triquet, C., Altmann, S., Amiard, J.C., Ballan-Dufrançais, C., Baumard, P., Budzinski, H., Crouzet, C., Garrigues, P., His, E., Jeantet, A.Y., Menasria, R., Mora, P., Mouneyrac, C., Narbonne, J.F., and Pavillon, J.F.	1998
8	EROD, MT	Metals and organic contaminants	Hylland, K., Nissen-Lie, T., Christensen, P.G., Sandvik, M., and Goksøyr, A.	1998
9	EROD, DNA adducts	Chrysene	Noaksson, E., Tjarnlund, U., Ericson, G., and Balk, L.	1998
10	FMO	Aldicarb	Schlenk, D., and El-Alfy, A.	1998
11	Haemolymph osmolality	Cu, As, BaP	Bamber, S.D., and Depledge, M.H.	1997
12	Interrenal cortisol and plasma lactate	Anoxia	Santos, M.A., and Pacheco, M.	1996
13	FMO activity (thiourea-*S*-oxidase)	None	Schlenk, D., Peters, L.D., and Livingstone, D.R.	1996
14	FMO, cytochrome P450	None	Schlenk, D., Peters, L.D., and Livingstone, D.R.	1996
15	Scope-for-growth	Cd	Guerin, J.L., and Stickle, W.B.	1995
16	P450, NADPH-cytochrome c reductase, catalase, SOD, GPX, DT-diaphorase	PAHs, PCBs, DDT, lindane	Sole, M., Porte, C., and Albaiges, J.	1995
17	FMO	None	Schlenk, D., Peters, L., Shehin-Johnson, S., Hines, R.N., and Livingstone, D.R.	1995
18	EROD	PCBs, PCDDs, PAHs	Eggens, M., and Bergman, A.	1995
19	MXR, MDR, non-specific esterase activity (FDA), AChE	PAHs, organophosphates	Minier, C.	1994
20	Haemolymph, haemocyanin	Cu	Weeks, J.M., Jensen, F.B., and Depledge, M.H.	1993
21	MT	Cd	Howard, C.L., and Hacker, C.S.	1990
22	MT	Zn	Baer, K.N., and Thomas, P.	1990
23	Glycerol production	Cu	Lustigman, B., McCormick, J.M., Dale, G., and McLaughlin, J.J.A.	1987
24	MT, HSP, cytochome P450	Municipal and industrial effluents, industrial chemicals, sewage, etc.	Kerambrun, P.	1984
25	ALA-D	Pb	Bouck, G.R.	1984

FDA: fluorescin diacetate; FMO: flavin-containing mono-oxygenase; GPX: glutathione peroxidase; GST: glutathione-*S*-transferase; HSP: heat shock protein(s); MDA: malonedialdehyde; MDR: multidrug resistance; MFO: mixed-function oxidase; MT: metallothionein; MXR: multixenobiotic resistance; SOD: superoxide dismutase.

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degree of control not often found when validating laboratory studies through field surveys. The converse may also be true in that an *in situ* assay may also be used to provide a degree of control and confirm measurements made in the field. In general, *in situ* assays are deployed for time periods of weeks or months, which to some extent reflects the effort required in deployment and retrieval of the experimental *in situ* system. A very important aspect of *in situ* methods is that the exact site location for a deployment can be precise, for example, close to point discharges.

Problems often encountered in field investigations are the lack of the species under study at the site of interest, and the mobility of the species. This applies especially to some fish species, e.g., flounder and sticklebacks in estuaries. In an *in situ* bioassay, the choice of species, uniformity in terms of origin (possibly genetic uniformity), size, age, sex, and reproductive state can be selected.

The main points of value of *in situ* bioassays are as follows:

1) they are particularly useful for field surveys with species not present naturally;
2) they also permit the use of standardized organisms with identical backgrounds;
3) they permit flexibility of deployment times and/or durations;
4) they can measure both biological effects and bioaccumulation;
5) biological endpoints can range from mortality through physiological changes, to biomarker responses;
6) they permit measurement of responses at specific locations;
7) exposure equipment can often be constructed from inexpensive materials, although this does not always apply;
8) they can allow field checking of laboratory bioassay results.

Some examples of the successful use of *in situ* bioassays in the marine environment include the following:

- In the UK EDMAR programme, which is studying endocrine disruptors, flounders (*Platichthys flesus*) and sticklebacks (*Gasterosteus aculeatus*) have been deployed in cages in estuaries, e.g., the Tees (Matthiessen *et al*., 2000). For flounders, the cages are constructed of steel and measure 1.5 m × 0.5 m × 0.5 m. Each cage is suitable for approximately thirty 20 cm-size fish. The cages sit on the seabed and the weight of the cage is sufficient to bed the cage into the mud; this is preferred since flounders naturally bury into the seabed. In the winter when the sea temperature is below 10 °C, the fish will survive without food for between four to six weeks. In the most recent study, the fish were removed from the cages and analysed for oestrogenic effects. In the stickleback studies, cages were constructed of a steel frame (0.6 m × 0.3 m × 0.3 m) covered in 5 mm nylon mesh. Each cage holds fifty fish of approximately 30 mm in size. The cages are deployed in a similar way to the flounder cages above, except that the cages must be suspended off the seabed. In these studies, the fish survived without food for ten weeks and were used for the detection of androgenic substances. Stickleback eggs and adults have also been deployed in cages in Canada.
- There have been several studies in the UK that have used *in situ* techniques with molluscs. *Crassostrea gigas* were used extensively in the 1980s for monitoring the effects of TBT (Waldock *et al*., 1987). Oyster spat (5 mm in size) were obtained from an oyster hatchery and placed in meshed trays located on the foreshore at mean low-water mark. Spat growth, shell thickening, and TBT bioaccumulation were measured in monthly samples taken over a ten-month period from March to December. This study highlighted the benefits of deploying *in situ* techniques: using organisms from the same origin and a uniform size; using organisms that did not occur naturally at the study sites; precise site selection.
- Mussels (*Mytilus edulis*) have been widely used in *in situ* deployments in tidal and subtidal locations in Europe. The cages are usually constructed of plastic mesh formed into a tray or cylinder and may contain up to 200 mussels. The cages in intertidal deployments are attached to piers, trestles or concrete blocks at mean low-water mark. The mussels in these studies can be used for the measurement of scope-for-growth (SFG) and somatic growth (meat, shell growth, and condition), as well as several other endpoints, and the deployments can be for many months. For offshore deployment, reinforced cylindrical meshed cages are constructed and are anchored and buoyed in position (Roddie and Johnson, 1990). In one instance, offshore deployments were made by suspending cages from light vessels located along the east coast of England (Widdows *et al*., 1995). In the two above-mentioned deployments, the cages were deployed for up to eight weeks, in the first instance to measure SFG at a sewage sludge disposal site and in the latter to measure SFG as part of an extensive study on the east coast of the UK. Bivalve molluscs have also been used in North American *in situ* deployments (Salazar and Salazar, 1991, 1996).
- Imposex in the mollusc *Nucella lapillus* held in mesh cages has been used to assess the spatial impact of TBT. Extensive studies in the mid-1980s revealed that some populations had died out in areas of high TBT contamination (Gibbs *et al*., 1987). In such instances, transplants of *N. lapillus* were made at these sites using cages. Approximately 100 animals

were placed in cages containing mussels as a food source for periods of up to twelve months, after which the animals were used for the determination of imposex and the analysis of TBT in the tissues.

- Kocan and Landolt (1990) have developed an *in situ* bioassay using herring (*Clupea harengus*) eggs. The eggs are sticky and normally adhere to rocks in nearshore areas. This adhesive quality is used for experimental purposes, because the eggs can be spawned directly onto an artificial substrate such as a glass plate. These plates can then be placed at specific locations in the field and a variety of endpoints determined, for example, embryo mortality, chromosome damage, hatching success, and teratogenic rates. The only drawback with this technique is that it is limited to the seasonal availability of spawning herring.

- Pollution-induced community tolerance (PICT) in marine plankton is an *in situ* technique that has been developed by Blanck and co-workers (1988). Periphyton is sampled using an artificial substrate, which consists of circular glass discs (1.5 cm^2) mounted on polyethylene holders, suspended 1.5 m below the water surface for 2–4 weeks. Each sampler holds up to 170 glass discs. Those specimens and species that are able to tolerate the conditions and thrive at the particular site of deployment colonize each disc. Sensitive specimens affected by the presence of contaminants are not able to compete and are therefore not represented on the sampling discs. Species diversity and photosynthetic rate are the endpoints used. The method has been used successfully to show PICT in a gradient of TBT contamination (Blanck and Dahl, 1996).

- Ten-day acute sediment toxicity tests using amphipods are now used widely in North America and Europe for regulatory and monitoring purposes. In one programme in the Netherlands (RIKZ), *Corophium volutator* has been used to measure the toxicity of dredge spoils for disposal at sea. Research at RIKZ has investigated and compared the use of the standard ten-day whole sediment test with an *in situ* method. In the *in situ* method, a frame is placed on the seabed at the site where dredge spoil is to be removed. A meshed plastic cylinder is secured onto the frame such that the cylindrical part is embedded into the sediment and the meshed part is in contact with the overlying water. Up to twenty *C. volutator* are placed in each cylinder using a special syringe. After ten days, the meshed cylinders and mud into which they have been placed are removed and taken to the laboratory for sieving and assessment of animal survival. In most cases, the *in situ* response was significantly more sensitive than the response in the laboratory. The reasons for this are still being investigated.

- *In situ* sediment bioassays have been developed in the USA by Tagatz and Deans (1983), in the UK by Matthiessen and Thain (1989), and in Norway by Olsgard (1999) for studying the effects of contaminated sediments on colonizing intertidal and subtidal organisms. In the study by Matthiessen and Thain, for example, clean natural sediments were removed from an intertidal shore, defaunated by repeated freezing, and spiked with TBT and a diesel-based drilling mud. The contaminated sediments were returned to the intertidal shore and the ability of animals (in the overlying water and surrounding clean sediment) to colonize the contaminated sediments was measured over a period of five months.

These and other studies have shown the value of *in situ* bioassays for marine monitoring, but have highlighted a number of important factors, which must be taken into account during their use:

1) It can be important and even essential to control for such factors as food availability, temperature, shading, salinity, and organism density;
2) It cannot be assumed that the exposure of caged organisms is necessarily the same as for adjacent wild organisms;
3) Cages may require disguise to prevent theft, or strong anchoring to prevent loss from currents, wave action or fishing activities;
4) Fouling of cages can be a confounding factor (e.g., through competition for food) although for some species (e.g., sticklebacks) the fouling organisms may provide a useful food source;
5) It is important to ensure that cage materials are non-toxic and inert (e.g., if studying endocrine-disrupting chemicals (EDCs), avoid plastics containing plasticizers);
6) Control mortality can be high in some species, and in some experimental systems it may be essential to exclude predators;
7) Appropriate acclimation and transport of test species is critical.

Recommendations

It is clear that careful use of *in situ* bioassays, alongside more traditional measurements in wild organisms and in laboratory-based bioassays of water and sediments, can add value to marine environmental monitoring programmes. ICES therefore recommends to Member Countries not already using *in situ* bioassays that consideration be given to the incorporation of such techniques in their marine monitoring strategies. International monitoring programmes such as those organized by OSPAR and HELCOM could also benefit from the inclusion of certain *in situ* bioassays (e.g., various whole-organism and biomarker endpoints in caged blue mussels (*Mytilus edulis*)) which are capable of successful deployment over much of the ICES area.

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6.1.3 Techniques to measure PAH metabolites

Request

This is part of the continuing ICES work to provide advice on method developments to be used in monitoring biological effects of contaminants in the marine environment.

Source of the information presented

The 2000 report of the Marine Chemistry Working Group (MCWG) and ACME deliberations.

Status/background information

Finfish do not generally accumulate high concentrations of polycyclic aromatic hydrocarbons (PAHs) as they possess an effective mixed-function oxygenase (MFO) system, which allows them to metabolize PAHs and to excrete them in the bile. An assessment of the exposure of fish to PAHs therefore also requires the determination of PAH metabolite concentrations in bile samples, as turnover times can be extremely rapid. Several methods are currently used to determine PAH metabolites. The simplest approach is to employ 1-hydroxypyrene as a marker of PAH exposure and use synchronous fluorescence spectrometry for quantification. A more complicated approach is to determine a range of PAH metabolites using HPLC with fluorescence detection. See the references below for more details. Methods for the measurement of PAH metabolites are, however, still under development. A European project is currently under way which aims to develop a fish oil reference material certified for PAH metabolites. During the course of that project, an intercomparison exercise has been conducted, and development of the reference material has begun. This has involved the preparation of contaminated fish oil, and tests to study and prevent the oxidation of PAH metabolites in these matrices. Standard solutions have also been prepared and are being tested for stability. As the results of this project are still emerging, it is too early to make a firm recommendation concerning techniques to measure PAH metabolites.

Need for further research or additional data

There is a need to obtain more information before any recommendations concerning techniques to measure PAH metabolites can be made.

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6.1.4 Sea-Going Workshop on the Effects of Contaminants in Pelagic Ecosystems

Request

This is part of continuing ICES work to improve the tools available for monitoring the biological effects of contaminants in the marine environment.

Source of the information presented

The 2000 report of the Working Group on Biological Effects of Contaminants (WGBEC) and ACME deliberations.

Status/background information

The ACME noted that WGBEC has been developing plans for a sea-going workshop concerning methods for studying effects of contaminants in pelagic organisms. This follows earlier sea-going workshops that have focused mainly on methods for evaluating effects in sediment dwellers. The objective of the workshop is to bring together scientists in order to assess the ability of selected uniform and standardized methods to detect biological effects of contaminants in pelagic ecosystems. The methods will be assessed for their applicability in future monitoring programmes. The workshop will use both field-collected organisms and organisms kept *in situ* in cages. The organisms expected to be available through field collection and in cages are indicated in Table 6.1.4.1.

Table 6.1.4.1. Organisms expected to be available for the sea-going workshop. In cages: Atlantic cod, blue mussel, herring eggs, stickleback. Wild organisms: see below. In addition: water, phytoplankton, microzooplankton, and bacteria will be available at all locations and during all periods.

Period	German Bight	East Shetland Basin	Reference areas
February 2001	Ripe dab, ripe whiting, dab eggs/embryos, plaice eggs/embryos, whiting eggs/embryos, sprat eggs/embryos, herring juveniles, overwintering *Calanus*	Fish eggs/embryos, juvenile herring, overwintering *Calanus*	Fish eggs/embryos, juvenile herring, overwintering *Calanus*
March 2001	Dab eggs/embryos, plaice eggs/embryos, whiting eggs/embryos, sprat eggs/embryos, juvenile herring	Ripe whiting, fish eggs/embryos, juvenile herring	Ripe whiting, fish eggs/embryos, juvenile herring
April/May* 2001	Mackerel eggs/embryos, fish larvae, juvenile herring, copepods	Mackerel eggs/embryos, fish larvae, juvenile herring, copepods	Fish larvae, mackerel eggs/embryos, juvenile herring
June** 2001	Fish larvae, juvenile herring, adult horse mackerel, copepods, surface microlayer	Fish larvae, juvenile herring, adult horse mackerel (possibly), copepods, surface microlayer	Fish larvae, juvenile herring, adult horse mackerel (possibly), copepods, surface microlayer
August 2001	Fish larvae, juvenile herring, adult horse mackerel, copepods	Fish larvae, juvenile herring, adult horse mackerel (possibly), copepods	Fish larvae, juvenile herring, adult horse mackerel (possibly), copepods

*cages deployed and one sampling (10-day exposure), **cages collected

The workshop is scheduled to take place in spring 2001. In February and May 2000, meetings of the scientific steering group were held under the chairmanship of Dr K. Hylland (Norway). A general strategy has been developed and a Prospectus prepared. The Prospectus was distributed to many European (and some North American) laboratories and institutions, and more than thirty individual project proposals have been received from scientists in ten countries. These proposals cover a broad range of techniques, including bioassays of water and solvent extracts using various test organisms, and biomarker and physiological measurements in wild and caged organisms. A final programme for the workshop is being established following evaluation of these proposals.

Various countries have placed research vessels at the provisional disposal of the workshop, e.g., England and Wales (RV "Cirolana"), Germany (RV "Walther Herwig"), Norway (RV "Michael Sars"), and Scotland (RV "Scotia"). Cages will be constructed at the Institute of Marine Research in Bergen, Norway. Furthermore, laboratories will be made available for post-cruise laboratory exposures in Bergen (Norway), Burnham-on-Crouch (England), Aberdeen (Scotland), and Kamperland (the Netherlands). A detailed budget for the workshop will be prepared before the ICES Annual Science Conference in September 2000.

Two areas of known contamination were selected, the German Bight and the East Shetland Basin area. In addition, relevant reference areas in the North Sea have also been identified. A preliminary schedule for the workshop has been established (Table 6.1.4.2).

Table 6.1.4.2. Preliminary schedule for workshop activities.

Task	Schedule
Meeting of extended group	3 February 2000
Prospectus distributed	February to April 2000
Deadline for proposals	1 May 2000
Evaluation – final programme	26 May 2000
Introductory meeting	February 2001
Cruises, practical work	February to September 2001
Wrap-up workshop	Planned for September 2002

6.2 HELCOM Workshop on Background/Reference Values for Concentrations of Nutrients and Chemical Contaminants in the Baltic Sea

Request

Item 4 of the 2000 requests from the Helsinki Commission: to participate in the planning and carrying out of the proposed workshop on background/reference values for concentrations of nutrients and hazardous substances in the Baltic Sea area, in the year 2000.

Source of the information presented

The 2000 report of the Marine Chemistry Working Group (MCWG) and ACME deliberations.

Status/background information

The ACME considered the HELCOM request for ICES to participate in the planning and conduct of a workshop on background/reference values for concentrations of nutrients and chemical contaminants in the Baltic Sea area, with the aim of improving the assessment work and harmonization with OSPAR activities. It was noted that background/reference values can also serve as reference values for criteria in matters relating to activities such as the dumping of dredged materials, and help to describe what can be considered as a healthy environment to the broader public. The ACME noted that it could also be valuable to compare the concentrations found in the environment with other types of limits, e.g., limits based on effects levels such as the ecotoxicological assessment criteria developed within OSPAR.

Of special interest for HELCOM are recommendations concerning the methods that should be used to define background concentrations and guidance on how these values should be derived. A number of approaches are currently suggested, such as the use of historical data for nutrients, and data from deep sediment cores to establish historic concentrations of contaminants in sediments.

In its discussion of this request, MCWG referred to comments that it had made in 1995 on the report of the first OSPAR workshop on background concentrations dealing with similar problems in the OSPAR area, and felt that these comments are still relevant. Some of the comments of a more general nature which could be of interest for the HELCOM workshop are repeated here (see also ICES, 1995).

There generally are considerable problems to determine background concentrations for naturally occurring compounds in matrices such as sea water, which change with time and which leave little or no record of their previous composition. By seeking to represent background conditions such as those which existed before 1950, direct analysis of sea water is not available. The alternative, but rather different, approach of seeking to determine typical values in pristine areas also has inherent problems. Firstly, it is debatable if it is possible to find a truly pristine area and, secondly, there could be a large geographical variability that should be taken into account.

Background values for naturally occurring compounds are in general highly dependent on natural processes in the area, e.g., different geological, chemical, and biochemical processes including natural variations in rock compositions. Therefore, background concentrations should not be given as an average value but as a range for these types of compounds. It should also be stated clearly how and from which locations the different values have been derived.

In 1997, MCWG thoroughly reviewed the report from the Second OSPAR/ICES Workshop on Background/ Reference Concentrations and these comments were also found to still be relevant. It was noted that the Second OSPAR/ICES Workshop on Background/Reference Concentrations had taken a different, more pragmatic approach to this problem by not distinguishing between these two types of values. Some of the comments made in 1997 of a more general nature, which could be useful for the planned HELCOM workshop, are given below (see also ICES, 1997).

It could be argued that for man-made compounds such as PCBs the background concentration is zero. If this is not acceptable, then it is better to speak of present minimum concentration values in surface sediments and in biota, due to the long-range atmospheric transport of, e.g., PCBs.

In comparing data with background values, uncertainties must be taken into account, both the analytical variance as well as the uncertainty of the boundaries of the background ranges. Errors due to sampling, handling, and analysis can always occur owing to contamination problems when samples contain low concentrations. This can influence the concentration ranges in areas often thought of being pristine.

Concerning the proposed HELCOM workshop for the Baltic Sea area, the ACME provided the following advice:

1) It is important to clearly define the terms "background concentration" and "reference concentration", respectively, taking into account that the term "background concentration" has different meanings in different disciplines.

2) It is important that the report from the workshop clearly explains how the different values have been derived.

3) Item number 4 in the draft Terms of Reference for the workshop ("to consider relevant OSPAR work in the field of historical background values") could be misleading if concentration values derived from the OSPAR workshops are to be considered for the Baltic Sea area, as those concentrations are not relevant for the Baltic Sea. However, the methodology from the OSPAR workshops should be considered in relation to the proposed HELCOM workshop.

4) Concerning nutrients, MCWG concluded already in 1997 that "Physical processes, such as upwelling in coastal areas, can displace large water volumes and cause rapid changes in the nutrient concentrations by bringing in, e.g., Atlantic water; consequently, the natural range of nutrient concentrations in a certain area can be quite large and include high concentrations. Enrichment by nutrients from anthropogenic sources will not necessarily make significant changes to the concentration range for the nutrients. Consequently, the use of so-called "background concentrations" would be misleading since they will not reveal any changes caused by anthropogenic sources." The principles described in this statement should be applicable also for the Baltic Sea.

5) When considering metals in sediment cores, the composition of the sediment samples must be taken into account.

6) As real background values, defined as prehistoric concentrations, for trace metals in sea water are probably something that can never be obtained, it is suggested that the present-day winter concentrations found in the Baltic relatively far from pollution sources can be used as practical reference values for the Baltic area for trace metals in sea water. However, it is important to remember that the sea water concentrations are always dependent on season, area, depth, salinity, and the presence or absence of oxygen. For example, it is well known that trace metal concentrations have a seasonal variability due to their affinity to be absorbed by and to adsorb to algae. Winter concentrations are therefore suggested as practical reference values, as the influence of algae is then expected to be minimized.

Need for further research or additional data

The ACME agreed that further development of assessment tools for concentrations of nutrients and chemical contaminants in the marine environment was necessary.

References

ICES. 1995. Report of the ICES Advisory Committee on the Marine Environment, 1995. ICES Cooperative Research Report, 212: 59–61.

ICES. 1997. Report of the ICES Advisory Committee on the Marine Environment, 1997. ICES Cooperative Research Report, 222: 107.

6.3 Arctic Monitoring and Assessment Programme: Developments in Monitoring and Assessment Activities

Request

This is ongoing work in cooperation with the Arctic Monitoring and Assessment Programme (AMAP).

Source of the information presented

Progress report from the ICES member of the AMAP Assessment Steering Group (Dr H. Loeng) and ACME deliberations.

Status/background information

The ACME noted that the Arctic Monitoring and Assessment Programme (AMAP) has finalized its Strategic Plan for the period up to 2003. The Ministers have focused AMAP activities for this period on the following priorities:

1) contaminant levels, trends, and effects in human populations and in the environment;
2) effects due to changes in climate and UV radiation;
3) source-receptor relationships;
4) human health;
5) communication of information.

AMAP monitoring activities are based, to the greatest extent possible, on ongoing national and international monitoring and research, aiming to harmonize this work and, where necessary, to promote new activities to fill identified gaps in order to meet the AMAP objectives. Certain projects of circumpolar importance, which are beyond national implementation plans, may require international steering and financing mechanisms. Close cooperation with other relevant regional and global programmes and observation networks, including data sharing and implementation of joint projects, is essential to avoid duplication of activities, to ensure optimal use of available resources, and to fill gaps in fundamental scientific knowledge needed for realizing AMAP's objectives.

The AMAP Trends and Effects Programme is the follow up of the Strategic Plan; it addresses the issue of data acquisition from two points of view:

- from the perspective of assessment concerning different pollutant/contaminant issues: determination of data and information needs for assessment of pollution issues and their effects on ecosystems and human health, including effects due to climate change and UV radiation;
- from the perspective of monitoring different environmental and ecosystem compartments: design of distinct sub-programmes for monitoring activities/stations, special field studies and other projects to cover data and information needs for future assessments according to the various environmental compartments concerned (atmospheric, marine, terrestrial and freshwater compartments, and humans with respect to human health).

The programme is rather comprehensive and includes information on station network and sampling strategies, supporting studies, and data reporting. Tables give detailed information on the media and parameters to be monitored, and organisms to be sampled for effect studies.

AMAP plans to complete its report on persistent organic pollutants (POPs), heavy metals, and radioactivity in autumn 2002. An assessment report on oil will be ready in 2004 and a report on acidification in 2006. Due to the fact that the expert/drafting groups for POPs and metals, in particular, have access to only a limited number of experts on biological effects, and generally the same individuals have the expertise relevant to both groups, establishment of an independent biological effects group is proposed. This group will support the work conducted by the other contaminant issue groups.

AMAP has already organized several workshops that have produced reports (see reference list) and several others will be organized throughout the assessment process.

In the spring of 1999, AMAP, CAFF (Conservation of Arctic Flora and Fauna), IASC (International Arctic Science Committee), IPCC (Intergovernmental Panel on Climate Change), and WCRP (World Climate Research Programme) jointly explored the idea of preparing an assessment of climate change and its consequences with leaders in science, government, and other interested bodies. These explorations and detailed discussions led to a formal proposal to the Senior Arctic Officials of the Arctic Council to plan for and conduct an Arctic Climate Impact Assessment (ACIA), including the effects of increased UV radiation, over the next several years.

The goals of ACIA are to:

a) evaluate and synthesize knowledge on climate variability, climate change, and increased UV radiation and their consequences; and

b) provide useful and reliable information to the governments, organizations, and peoples of the Arctic region in order to support policy-making processes.

The assessment will include environmental, human health, and social and economic impacts, and recommendations for further actions. The assessment will be conducted in the context of other developments and pressures on the Arctic environment, its economy, regional resources, and peoples.

ICES was invited to have a member on the AMAP Assessment Steering Committee (ASC), and until June 2000, the Chair of the ICES Oceanography Committee has been that member. Drafting groups led by a lead author appointed by the ASC will carry out the writing of the ACIA. The first report will be ready in 2004.

Recommendation

ICES should continue to be involved in the work of preparing an Arctic Climate Impact Assessment Report (ACIA), and should continue to participate as a member of the AMAP Assessment Steering Committee (ASC).

References

AMAP. 1999a. Modelling and Sources: A Workshop on Techniques and Associated Uncertainties in Quantifying the Origin and Long-Range Transport of Contaminants to the Arctic, Bergen, Norway. AMAP Report 99:4. Arctic Monitoring and Assessment Programme, Oslo.

AMAP. 1999b. Minutes from the Thirteenth Meeting of the Arctic Monitoring and Assessment Programme Working Group (AMAPWG), Toronto, Canada, 10–12 November 1999. AMAP Report 99:5. Arctic Monitoring and Assessment Programme, Oslo.

AMAP. 1999c. AMAP Strategic Plan: 1998–2003. AMAP Report 99:6. Arctic Monitoring and Assessment Programme, Oslo.

AMAP. 1999d. The AMAP Trends and Effects Programme. AMAP Report 99:7. Arctic Monitoring and Assessment Programme, Oslo.

AMAP. 1999e. Heavy Metals in the Arctic. Anchorage, Alaska, 7–10 September 1999. Proceedings. AMAP Report 99:8. Arctic Monitoring and Assessment Programme, Oslo.

AMAP. 2000. International Workshop on Persistent Organic Pollutants (POPs) in the Arctic: Human Health and Environmental Concerns, Rovaniemi, Finland, 18–20 January 2000. Proceedings. AMAP Report 2000:1. Arctic Monitoring and Assessment Programme, Oslo.

6.4 Substances (Nutrients, Organic Contaminants, and Trace Elements) in Marine Media that can be Monitored on a Routine Basis

Request

This updates information presented in previous ACME reports and is of interest to organizations coordinating international or regional monitoring programmes on nutrients and contaminants in marine media.

Source of the information presented

The 2000 report of the Marine Chemistry Working Group (MCWG) and ACME deliberations.

Status/background information

In its 1999 report, the ACME presented comprehensive tables on the performance of all laboratories that had taken part in the QUASIMEME Laboratory Performance Scheme (LPS). The tables provided a summary of results from ten or eleven exercises carried out over 2.5 years and were used as indicators of the ability of laboratories to perform routine monitoring (ICES, 2000).

At its 2000 meeting, MCWG examined new tables from the QUASIMEME LPS with results from an additional year and concluded that there was not much difference in performance from year to year and that an assessment of overall laboratory performances would be sufficient every three years.

However, with respect to trace metals the lowest concentrations in sea water, sediments, and biota that can be monitored on a routine basis by a group of laboratories were estimated, based on the percentage of satisfactory results in the QUASIMEME LPS (Table 6.4.1). There, a satisfactory result for a group performance was defined on the criterion that the majority (here > 60 %) of laboratories received Z-scores between $-2 < Z < 2$ for an individual parameter/matrix combination. A decreasing number of successful laboratories with decreasing analyte concentration is apparent.

Furthermore, MCWG discussed ways of presenting and assessing information on laboratory performance and suggested that instead of summarizing the performance

Table 6.4.1. Lowest concentrations of trace elements in sea water, sediments, and biota which can be monitored on a routine basis by the majority of laboratories (according to the percentage of satisfactory results in the QUASIMEME LPS, 1996–1999; for sea water, 1998–1999 only).

Trace element	Sea water ($\mu g\ l^{-1}$)	Sediments ($mg\ kg^{-1}$ dw)	Biota ($mg\ kg^{-1}$ ww)
Zn	7	75	≤ 4.6
Cd	≤ 0.09	≤ 0.011	Fish tissue: 0.005
Pb	0.5	40	Problems for the majority of the labs
Cu	≤ 0.9	17	≤ 0.26
Cr	2	28	≤ 0.14
Ni	≤ 14	23	For cod liver and muscle: 0.1
As	≤ 1.2	6	≤ 1.3
Hg	0.007	0.12	≤ 0.028
Se			0.4
Al		Value not available*	
Mn		≤ 750	
Fe		≤ 2.8 %	
Li		≤ 35	
Sc		≤ 7.6	

"≤" means that only a less than concentration can be given and not a minimum concentration for which more than 60 % of the laboratories achieved satisfactory results ($-2 < Z < 2$). A minimum concentration could not be given from the results of the QUASIMEME LPS, as the concentrations of the samples used were not low enough.

*Method dependent. Some laboratories do not use hydrofluoric acid for complete dissolution of the sample. There is generally no detection limit problem for aluminium.

of all laboratories participating in the QUASIMEME LPS, other useful alternative categories were:

- only laboratories in ICES Member Countries;
- laboratories performing analyses for specific monitoring programmes, e.g., HELCOM, OSPAR, and MEDPOL, so that data assessment groups can easily include information on the comparability and reliability of routine monitoring data in their assessment.

With data in these categories at hand, the lists of mandatory parameters in monitoring programmes can be evaluated based on information about which parameters can be monitored on a routine basis.

Need for further research or additional data

Evaluation of the data in the QUASIMEME LPS time series may provide useful information on the capability of laboratories working in specific regions or monitoring programmes. Similar data would be useful on monitoring laboratories which are not in the QUASIMEME LPS, e.g., from the west side of the Atlantic.

Reference

ICES. 2000. Report of the ICES Advisory Committee on the Marine Environment, 1999. ICES Cooperative Research Report, 239: 33–38.

6.5 Techniques for Sediment Monitoring

6.5.1 Normalization techniques for contaminant concentrations in sediments

Request

This topic is part of ongoing ICES work on the monitoring of contaminants and is of relevance for the OSPAR Commission.

Source of the information presented

The 2000 report of the Working Group on Marine Sediments in Relation to Pollution (WGMS) and ACME deliberations.

Status/background information

For several years, tremendous efforts have been directed at the development of new techniques for normalizing concentrations of contaminants in sediments. The ongoing implementation of the OSPAR Coordinated Environmental Monitoring Programme (CEMP) that stipulates temporal trend and spatial surveys of contaminants in sediments on a continuous and periodic basis, respectively, encouraged the ACME to review currently existing viewpoints and new insights on normalization techniques. The outcome of the 2000 meeting of WGMS on this topic formed the basis for the ACME's review and deliberations. The contribution of WGMS is attached to this report as Annex 1.

The ACME agreed that the annex clearly reflects the current insights on normalization for spatial and temporal trend surveys. The ACME stated that the further improvement of this text could provide a sound basis for robust guidelines that can contribute with a reasonable added value to the existing guidelines. The ACME recommended that earlier ACME advice (ICES, 1994a, 1994b, 1995, 1996, 1997) and the 2000 WGMS report should also be considered in this process.

In order to ensure a successful application of proposed normalization techniques on a broad scale, e.g., the OSPAR area, several clarifications on uncertainties in the required knowledge are needed. It is therefore advised that the following comments and views of the ACME are taken into account.

The ACME noted that, apart from the aim of comparing concentrations in sediments from different areas in spatial studies, normalization primarily aims to account for the natural variability (e.g., the naturally occurring metal contents) in particle size distribution and mineralogy and distinguish it from contributions from anthropogenic sources. The current monitoring guidelines include normalization methods on analysis of whole sediments (< 2 mm) for both metals and organic compounds (by total digestion/extraction methods) that have been successfully applied in Canada, the USA, and Europe. This work has been widely documented in the scientific literature. The value of these procedures is well recognized, and the text in Annex 1 indicates the strengths and weaknesses of these approaches. Weaknesses include, for example, the low concentrations and high uncertainties in analyses of whole sandy sediments and the consequential difficulties in incorporating data from sandy sediments in contaminant distribution maps. The scientific challenge is to improve on what has been used previously.

It is recognized that the determination of total concentrations of contaminants in whole sediment is the closest existing approach to provide a direct chemical measurement giving a direct description of the seabed sediment. However, considerable limitations on the utility of these data have been found in recent attempts to compare contaminant concentrations in spatial studies covering large geographical areas, such as the whole OSPAR area. While chemical analysis of sieved fractions does not reflect the situation of the whole sediment, it has been shown in the North Sea area (in association with a second normalization to co-factors such as aluminium or carbon) to greatly reduce the variance of data and allow more reliable comparisons to be made between locations and between sampling occasions.

The further normalization of the sieved fraction data, as proposed by WGMS, requires the establishment of statistical and analytical procedures to assess the overall effectiveness of this normalization as, to the knowledge of ACME, this has never been applied to a geographical area wider than the North Sea, such as the OSPAR area. For the same reason, the reliability of maps on spatial distributions of contaminants, based on this extended normalization, should be assessed. The production of these maps will require additional mapping of co-factors.

The ACME endorsed the clear view of WGMS not to recommend partial digestion for metal determinations on a broad geographical scale. Partial digestion refers to digestion methods that do not result in total dissolution of the sediment and which, therefore, may not measure the total contaminant or normalizer content. More research is needed on the standardization and applicability of partial digestion. The data sets considered by WGMS were restricted to Cd, Pb, Cu, and Zn. Certified reference materials (CRMs) are not available for partial digestion purposes. Furthermore, the degree of decomposition depends largely on the mineralogical composition of the sediment. Differences in mineralogy between samples may require alterations to be made to the digestion procedures to obtain "comparable" degrees of decomposition. The ACME therefore concluded that more information on partial digestion should be gathered, and this work should be extended to include other elements.

The ACME also felt the need to highlight its concern that the growing complexity of the normalization tools might endanger their applicability in monitoring programmes. Annex 1 was therefore particularly helpful in clarifying the range of normalization tools that are available, and the advantages and limitations of each. Agencies planning sediment analytical programmes could balance the additional effort required to analyse sieved sediments with the improved comparability of the resulting data. On the other hand, there are indications that, for some dynamic coastal areas in the western European Atlantic, e.g., west of the UK, Ireland, Norway, Bay of Biscay, and the Iberian coast, application of the sieved sediment approach may be misleading.

The ACME noted that sediment monitoring programmes now often sought to integrate sediment chemistry with sediment toxicity assessment. It is not clear how sediment analyses of sieved material could be related to toxicity assessments which are commonly carried out on whole (< 2 mm) sediment.

Sieving concentrates the fine particles, and the associated metals as well as the organic matter, together with the associated organic compounds. Expression of the sieved sediment analyses as ratios to appropriate co-factors may be helpful in integrating the two types of data.

It is obvious that temporal trend analysis requires adequate normalization. Normalization for temporal trends should not necessarily be based on internationally agreed co-factors. The criteria for suitable co-factors should be balanced on the following site-specific requirements: 1) they should sufficiently minimize the natural variabilities; 2) they should be accurately measurable; and 3) they should be cost effective.

Need for further research or additional data

In spite of the efforts of WGMS, finalized guidelines on normalization of contaminants in sediments are still not available. However, combining Annex 1 of this report with the relevant material from the 2000 WGMS report, and taking account of previous ACME statements, should provide a suitable document.

There is a requirement to establish statistical and analytical procedures to assess the overall effectiveness of the normalization of data from sieved fractions over a wide geographical area such as the OSPAR region.

Recommendations

ICES recommends that the existing temporal trend programmes should continue on the basis of the established designs, provided that the assessment of the data indicates that the statistical power of the programmes is adequate for the overall objectives. New temporal trend programmes should be carried out by the analysis of a fine-grained fraction, isolated by sieving if necessary.

In addition, ICES recommends that:

a) if the purpose of sediment monitoring is to establish the existence or otherwise of temporal trends in contaminants, then normalization based on analysis of the fine fraction is an appropriate procedure, and should be employed in new temporal trend programmes;

b) further research is required into analytical and statistical procedures to be employed for normalization using the fine fraction, when spatial trends in contaminants over wide geographical areas are to be studied. In the meantime, wide-scale spatial trend monitoring should avoid normalization through analysis of the fine fraction;

c) on the other hand, if the purpose of monitoring is to predict or interpret the biological effects of sedimentary contaminants on benthos organisms, then analysis of total sediment (< 2 mm) is the preferred approach;

d) if it is desired to use a particular sediment sample for both temporal trend monitoring and for the prediction/interpretation of biological effects, then both whole sediment and fine fraction analyses should be conducted on that sample;

e) partial digestion of whole sediment for metal determinations should not be used on a broad geographical scale due to problems with detection of the total contaminant contents, the absence of CRMs, and differences in the mineralogy of the sediment samples.

References

ICES. 1994a. Report of the ICES Advisory Committee on the Marine Environment, 1993. ICES Cooperative Research Report, 198: 45–57.

ICES. 1994b. Report of the ICES Advisory Committee on the Marine Environment, 1994. ICES Cooperative Research Report, 204: 27–29.

ICES. 1995. Report of the ICES Advisory Committee on the Marine Environment, 1995. ICES Cooperative Research Report, 212: 33–34.

ICES. 1996. Report of the ICES Advisory Committee on the Marine Environment, 1996. ICES Cooperative Research Report, 217: 37–38.

ICES. 1997. Report of the ICES Advisory Committee on the Marine Environment, 1997. ICES Cooperative Research Report, 222: 25–26.

6.5.2 Organotin guidelines

Request

This is part of the continuing ICES work on the monitoring of contaminants in sediments.

Source of the information presented

The 2000 report of the Working Group on Marine Sediments in Relation to Pollution (WGMS) and ACME deliberations.

Status/background information

At its 1998 meeting, the ACME reviewed and adopted guidelines for the determination of tributyltin (TBT), including its metabolites dibutyltin (DBT) and monobutyltin (MBT), in sediments, prepared by WGMS, for inclusion as a Technical Annex to the overall Guidelines for the Use of Sediments in Marine Monitoring (ICES, 1994). These butyltin guidelines were included as Annex 2 of the 1998 ACME report (ICES, 1999) with a recommendation to OSPAR and HELCOM for their possible inclusion in the Joint Assessment and Monitoring Programme (JAMP) Guidelines and the Baltic Monitoring Programme Manual, respectively. The guidelines were adopted by OSPAR in 1999 (ASMO(1) 99/4/5). Already in 1998 the ACME felt that inclusion in the Technical Annex of procedures for the measurement of triphenyltin (TPT) and its metabolites, diphenyltin (DPT) and monophenyltin (MPT), would be appropriate.

The ACME reviewed and discussed a supplementary note prepared by WGMS at its 2000 meeting in response to this suggestion and to some comments made on the possible need for clarification of the use of different types of internal standards for calibration.

In this technical note, WGMS recognized the interest in developing analytical methods for phenyltin compounds in marine sediments. WGMS was informed that there is currently a European Commission Community Bureau of Reference (BCR) project under way to develop reference materials (RMs) for phenyltin analysis. According to the information, the project is making good progress on the determination of triphenyltin, but was experiencing considerable difficulty in the determination of diphenyltin and monophenyltin. WGMS therefore agreed that it is premature to prepare a detailed Technical Annex covering phenyltins. However, the situation should be reviewed once the current BCR project has been completed, with a view to preparing guidance, for example, in the form of an Addendum to the Technical Annex on TBT, or perhaps as a stand-alone document.

With respect to the calibration procedures, WGMS reviewed the current butyltin guidelines and agreed that the calibration procedures are suitably described. WGMS recognized that there may be other procedures that are equally effective, and also noted that the inclusion of the measurement of butyltins in OSPAR programmes would require the agreement of a Technical Annex and Quality Assurance procedures. Normally, the details of analytical Technical Annexes are not mandatory for all contributing laboratories; different methods can be used provided that the laboratories can demonstrate, for example, through participation in Laboratory Performance Studies, that their methods produce satisfactory results. In this way, it is possible to accommodate differences in calibration procedures between laboratories, as unsatisfactory differences would be apparent in the results, e.g., from the Laboratory Performance Studies. WGMS therefore did not consider it necessary to amend the guidelines on butyltin calibration.

A question was raised about the possibility of ICES preparing guidelines for the analysis of TBT and other organotin compounds in biota. It was noted that the Marine Chemistry Working Group (MCWG) briefly considered this issue at its 2000 meeting, but as relevant information which may significantly contribute to the development of such guidelines is expected to be available soon, MCWG decided to postpone work on this issue until this material can be examined.

References

ICES. 1994. Report of the ICES Advisory Committee on the Marine Environment, 1993. ICES Cooperative Research Report, 198: 45–57.

ICES. 1999. Report of the ICES Advisory Committee on the Marine Environment, 1998. ICES Cooperative Research Report, 233: 22–23.

6.6 Statistical Aspects of Monitoring

6.6.1 Development of trend detection methods for input data

Request

Item 2 of the 2000 Work Programme from the OSPAR Commission:

2 Completion of Work on Assessment Tools

2.1 Fine tuning of the Trend-Y-tector for trend detection in inputs

2.1.1 Consider type and specification of the Smoother. The current LOESS Smoother seems to be quite sensitive for outliers in the first and last part of the data series. ICES is asked to consider possible alternatives, e.g., the Spline Smoother.

2.1.2 Consider the calculation of residuals. Due to the risk that over-fitting of normal residuals may lead to underestimation of the standard deviation, ICES is asked if residuals based on cross validation may be a sensible alternative.

2.1.3 Consider the calculation of standard deviation. The current guidelines use the L moments for estimating the standard deviation. ICES is asked to consider the statistical consequences of using this or other robust estimators combined with corresponding critical values calculated by simulation studies.

2.1.4 Examine alternatives of the current smoother test. This test does not seem to be accurate. ICES is asked for possible alternatives e.g., splitting the trend into a linear and a non-linear component and then testing the linear part.

2.1.5 Establish the link between overall significance and individual significance levels.

2.1.6 Specify the procedure for power calculation, and especially the post hoc power as an integral part of the trend assessment.

2.2 Adjustment of loads

2.2.1 Consider the general procedure of adjustment and trend assessment as outlined in INPUT(2) 98/5/3 and 98/5/4 with regard to its statistical implications.

2.2.2 Consider and examine the choice of the statistical model and the underlying variables for adjustment.

2.2.3 Give advice with respect to the choice of multiplicative or additive adjustment.

2.2.4 Give advice on how to measure the gain of adjustment.

2.2.5 Examine whether and when there is a risk of "over-adjustment".

2.2.6 Consider whether the use of annual adjusted loads in the trend analysis of monthly loads may become redundant.

2.3 The use of monthly data

2.3.1 Development of provisions for the use of monthly data in these trend detection methods (taking into account that any recommendations should be based on real need and best scientific judgements and should not be driven purely by statistical considerations).

Source of the information presented

The 2000 report of the Working Group on Statistical Aspects of Environmental Monitoring (WGSAEM) and ACME deliberations.

Status/background information

Following the 1997 ICES/OSPAR Workshop on the Identification of Statistical Methods for Trend Detection, the 1997 and 1998 WGSAEM meetings had discussed several statistical issues concerning the analysis of data on inputs of nutrients and contaminants to the marine environment via rivers and atmospheric deposition. In 1999 and 2000, OSPAR requested further advice concerning three groups of questions, with priorities for answering them. These covered:

a) fine tuning of the Trend-y-tector (e.g., to develop a method for including a power function in the trend detection methods for input data);

b) adjustment of loads (to develop and assess statistical methods for dealing with data which are more complex than a series of independent, annual unadjusted loads; the methods should address adjustments to annual and monthly data for, *inter alia*, climatic effects);

c) the use of monthly data (to develop provisions for the use of monthly data in the trend detection methods).

Because of time restrictions, only items a) and b) were addressed in 1999 and the results were presented in Sections 6.2–6.4 of the 1999 ACME report (ICES, 2000). Sections 6.2 and 6.3 contained the responses to more detailed questions under a) and b), respectively. In Section 6.4, the ACME reported WGSAEM deliberations on what would be the desirable components of a package of trend assessment tools, based on lengthy experience with assessments of trends of contaminants in biota and limited experience for inputs, as well as their own expertise and following the general objective that the statistical method used to assess trends should be robust, intuitive, and revealing. Four separate but complementary components were identified:

1) graphical presentation of the time series with, for example, a summary line to indicate the general trend and tolerance lines to reveal potential data anomalies;

2) a formal test of trend, with trend defined in an appropriate way for the context of the assessment, and possibly with a power curve which reflects the detectability of the given trend;

3) a measure of the tendency to increase or decrease;

4) a comparison of the current level against some reference level or a level in a previous year.

Some of components 2), 3), and 4), but especially 3) and 4), might then be combined in a statistical meta-analysis to provide summaries across regions, or across contaminants, etc. These components were already included in the software application Trend-y-tector, but they were combined together into a single test of trend. Separating the components allows them to fulfill a more informative role, and also gives greater flexibility in choosing a statistical method for each component that is most appropriate for the purpose of a specific monitoring programme. It also makes it easier for the tools adopted for a specific programme to evolve to meet the particular needs of the assessment group. These needs are likely to be clarified with experience in using the package over a series of assessments.

In continuance of this work in 2000, WGSAEM was requested to develop further advice on trend assessment tools, in particular, "to continue the development of trend detection methods in order:

a) to consider further development and assessment of robust smoother methods and the development of appropriate techniques for revealing outlying data values;
b) to consider further development of statistical methods for adjustment of input loads;
c) to develop provisions for the use of monthly data in trend detection methods."

The OSPAR Working Group on Inputs to the Marine Environment (INPUT) had also agreed on further requests to be considered on a voluntary basis for advice on trend assessment tools. These consisted of:

d1) examination of the load adjustment procedure described in the current HARP Guideline 7 (OSPAR, 1999);
d2) the use of annual mean concentration instead of annual adjusted loads;
d3) if possible, providing some concrete examples of the application of the trend detection protocol;
d4) reviewing the Trend-y-tector;
d5) cooperating in writing draft guidelines for trend assessment with a wide scope of application, to be prepared by the Netherlands.

On behalf of the Netherlands, O. Swertz announced in a letter to WGSAEM that the Netherlands are intending to use the work of last year (Sections 6.2–6.4 from the 1999 ACME report) as a basis for preparing draft guidelines, with the aim of promoting the Trend-y-tector as an application for future OSPAR trend assessment guidelines. WGSAEM was requested to comment on that work and on the currently partly updated Trend-y-tector software application.

The following sub-sections summarize the WGSAEM work on this OSPAR request, as reviewed and accepted by the ACME.

Item a) Robust smoother methods

WGSAEM reviewed the decision of whether or not to use a robust smoother. In case that a robust smoother will be used, a standard robust smoother can be found in Cleveland (1993). As discussed in Section 6.4 of the 1999 ACME report, this should depend on the purpose of the assessment. If recent trends are intended to trigger a more detailed analysis and possibly further action, information about alarming recent trends may be given higher priority than making a statement unaffected by incidental outliers. In that case, a robust approach would not be recommended. The chance of being misled, if the most recent data contain outliers, is acknowledged.

In general, it is not useful to apply a robust smoother on aggregated data to control the influence of a single outlier within the year. If there are single outliers within a year, they should be detected and eliminated before producing an annual index. Alternatively, a robust statistic could be used for the annual index.

Item b) Adjustment of loads

The proposed procedure of adjustment, its interpretation and several adjustment methods are described in Annex 2, Part 1. It turns out that there is no one method that is optimal for every river and every substance. However, method L1 performs reasonably well in terms of smoothness of the annual adjusted load for the nutrients and partly also for heavy metals. Method L1 is based on a linear model for the concentration, with time and reciprocal flow as independent variables, together with a parametric seasonal component of second order. The calculation of the parameters is performed by local regression based on a running window of a given number of years. According to the model assumptions and the outcome of the statistical calculations, it can be concluded that adjustment according to method L1 performs reasonably well for nutrients and if the load flow relation is approximately linear. However, there are situations where other methods (e.g., method L2) perform better, e.g., when there is a significant effect of lagged runoff. Method L1 looks promising also for atmospheric depositions on a monthly basis and adjusted for precipitation, although other variables such as wind speed may also be adjusted for.

With regard to the difficulties to adjust the load of heavy metals in relation to the runoff, the ACME noted that heavy metals can interact with particles such as algae and other organic material as well as small clay mineral particles in the water phase. This may well have an effect on the concentrations of heavy metals in the dissolved fraction. An adjustment of the concentration data regarding particle concentration and composition might therefore be a way of improving the statistical methods for calculating trends in input loads. However, it has to be noted that the different metals react differently with particles, e.g., Pb and Hg have a much higher affinity for particles than, e.g., Cu. The composition of the particle should also be taken into account, as the metals are adsorbed to the different types of particles with different affinities and, in addition, some "particles" like algae can also actively absorb some metals.

Some reservations were expressed about using the "smoothness" in the time series of adjusted loads as the only criterion for choosing an adjustment method. Many of the adjustment methods used data from several adjacent years to construct each annual index, and this will induce correlations in the annual indices. It was unclear what impact this would have on statistical tests made using these indices.

WGSAEM therefore investigated the correlation structure induced by various adjustment methods, and its impact on inference, and the performance of some simpler adjustment methods. The flow data series from the Lobith/Rhine was used to calculate the covariance matrix of the annual (additive-) adjusted loads, assuming that there are 26 measurements each year, and assuming a window width of seven years.

A simulation showed that this resulted in a slight increase in the type I error of an F-test based on linear regression of the annual indices. Under a variety of flow regimes (ten-year snapshots from the Lobith/Rhine series), the type I error increased from 5 % to about 6 % at most. This is only a slight increase in type I error, so that for these data, and for the seven-year window, the annual adjusted loads behave almost like a stochastically independent time series.

Some alternative adjustment methods were considered that constructed the annual index using the data collected in that particular year. They had the great advantage of simplicity. Let (L_{ij}, q_{ij}), $j = 1, \ldots, m$, be the pairs of load and flow measurements in year i, let $\left(q_t^{(i)}\right), t = 1, \cdots, 365(366)$, denote the series of daily flows and $t_{i1}, t_{i2}, \ldots, t_{im}$ the sampling times in year i, i.e., $q_{ij} = q_{t_{ij}}^{(i)}$. Let Q be the mean flow over the entire time series. The annual indices were:

- Mean: the annual mean load $\overline{L}_i = \frac{1}{m} \sum_{j=1}^{m} q_{ij} c_{ij}$
- OSPAR: the annual load calculated according to the "OSPAR formula": $L_{i,OSPAR} = \dfrac{\frac{1}{365} \sum_{t=1}^{365} q_t^{(i)}}{\frac{1}{m} \sum_{j=1}^{m} q_{ij}} \overline{L}_i$.
- Ratio: $\overline{L}_i \times Q / \left(\frac{1}{m} \sum_{j=1}^{m} q_{ij} \right)$, also known as method A0 in Annex 2 and 1A1 in the HARP Protocol.
- Linear: the model $L_{ij} = \alpha_i + \beta_i \, q_{ij} + \varepsilon_{ij}$ is fitted assuming gamma errors and identity link: the annual index is then the predicted load at Q, $\hat{\alpha}_i + \hat{\beta}_i Q$.
- GAM: the model $L_{ij} = s_i(q_{ij}) + \varepsilon_{ij}$ is fitted assuming gamma errors and identity link, where $s_i(.)$ denotes a year-specific smoothing spline on two degrees of freedom that allows for a non-linear load-flow relationship: the annual index is then the predicted load at Q.

These indices were fitted to four time series: nitrate and total P in the rivers Rhine (Lobith) and Ems (Herbrum). To assess their performance, the residual standard deviation of the annual indices was calculated by fitting a LOWESS smoother with a span of eight years (and corrected for differences in the mean level). For comparison, the residual standard deviation was also calculated for the annual index L1 of Annex 2, Part 1, which uses information from adjacent years. The results (in tonnes per year) are shown in Table 6.6.1.1.

In terms of smoothness, method L1 performs best throughout. The GAM index also performs reasonably across all four time series. However, note that calculating the GAM index was sometimes difficult for the Ems River, where there were only twelve observations per year, due to convergence problems. It should also be noted that annual indices are not necessarily uncorrelated: if there are seasonal differences in the load-flow relationship, and there are also auto-correlations in annual flows, the GAM index might be auto-correlated as well.

WGSAEM concluded finally that at present no single method of calculating annual adjusted loads can be recommended for all river systems and all substances. However, when considering nutrients, if load is approximately linearly related to flow, method L1 will perform reasonably well and WGSAEM recommends that it should be used on a trial basis. The ACME endorsed this recommendation.

Item c) Analysis of monthly data

In order to obtain a better understanding of the advantages of analysing monthly data, several trend analysis methods were applied to a test data set. In Annex 2, Part 2, the different methods are described together with the protocols of the analyses.

Generally, analysing monthly data forces the incorporation of specific tools to treat serial correlations and seasonality, which makes the analysis much more complicated. Not accounting for serial correlation in monthly data may result in biased statistical tests due to serious violation of the statistical assumptions. This is illustrated in the calculations presented in Annex 2, Part 2. The use of one of these methods dealing with monthly data requires some statistical experience, and it is not clear at this stage whether it would be possible to define a method dealing with serial correlations that would be both routinely applicable to many data sets, and insensitive to statistical assumptions as well as to adverse numerical features such as extreme data values, partial bulking of samples, and measurements below detection levels. It should be noted that to some extent the problem of serial correlations is relevant also for annual data, however, WGSAEM felt that it can be neglected. Typically, auto-correlation between years is much smaller than between months. Apart from that, proper specification of the serial correlation is not possible on the basis of annual data from ten or even fewer years.

Considering these aspects, no general recommendation on the use of monthly data can be given. There is some theoretical evidence that analysing monthly data may be

Table 6.6.1.1. Results of constructing annual indices for four time series using six different methods.

	Rhine/Lobith		Ems/Herbrum	
	Nitrate	**Total P**	**Nitrate**	**Total P**
Mean	38085	3938	4347	267
OSPAR	37412	3544	3586	226
Ratio	11920	4952	1993	120
Linear	12808	3241	2879	152
GAM	13002	2757	2083	152
L1	9076	2545	1300	115

better to some extent in terms of trend detectability and also with regard to the actual significance level. On the other hand, the analysis of monthly data is much more complicated and it is not quite clear whether a method can be defined that can deal with many very different data sets.

Item d2) Use of annual mean concentration

WGSAEM noted that in case of seasonal variation, yearly averaged concentrations instead of annual loads or annual adjusted loads may not properly reflect the effectiveness of the reduction of inputs, not even in the long run.

In Table 6.6.1.2, a hypothetical example is presented that is based on the monthly mean runoff at Herbrum/Ems. The monthly mean runoff is given in the second column, and the third column contains the hypothetical nutrient mean concentration before reduction measures take place. The respective load is given in the fourth column, and the hypothetical mean concentration after the reduction measures become effective is contained in column 5, with the corresponding load in column 6. According to the figures given, the reduction measures are effective in reducing inputs by more than 40 %. This is obvious in the load (and would also be apparent in an adjusted load), with a reduction of 41.2 %, but due to a shift in the seasonal variation of the mean concentration, there is no reduction at all of the averaged concentration.

In order to avoid the risk of obtaining such a result, it was felt that the use of concentration mean values cannot be recommended.

Items d4) and d5) Reviewing the Trend-y-tector and draft guidelines

WGSAEM noted that the software application Trend-y-tector has been partly updated with the aim of fully complying with the ICES advice in Section 6.4 of the 1999 ACME report. WGSAEM discussed the intention of the Netherlands to use the work of last year as a basis for draft guidelines for trend assessment with a wide scope of application and noted that:

- Section 6.4 focused on the development of statistical methods that would be useful for large-scale assessments of very many data sets, where the emphasis was on producing standard assessments and reports for each time series. For small numbers of data sets, more efficient methods might be possible, taking account of the specific features of each data set.
- Section 6.4 had been created as a general guide to the development of trend assessment software, in particular to suggest features that should be considered in the development of the Trend-y-tector software. It does not provide guidelines for making use of such software or for the interpretation of results.
- Section 6.4 focuses on the assessment of input data. For example, assessing the target of a decrease of 50 % in a ten-year period is specific to OSPAR INPUT.
- Appropriate guidelines for the use of the Trend-y-tector software should be developed.

With regard to the Trend-y-tector software, it was noted that:

- for reference, the Trend-y-tector should be given version numbers;
- consideration should be given to providing more extensive additional analyses and reporting, such as % variation explained and, in the case of the smoother, further analysis and reporting of the linear and non-linear components of the trend in the whole time series;
- consideration should be given to extending the methods to allow more sophisticated treatment of missing values;
- in the current version, *post-hoc* assessment of power is not available.

Table 6.6.1.2. Monthly runoff at the Ems River and hypothetical nutrient concentrations before and after reduction measures.

Month	Runoff Herbrum (Ems)	Concentration	Load	Concentration	Load
	Monthly mean ($m^3 s^{-1}$)	Before ($mg l^{-1}$)	Before ($g s^{-1}$)	After ($mg l^{-1}$)	After ($g s^{-1}$)
1	183.2	1.5	274.8	0..2	36.6
2	154.5	1.8	278.1	0.2	30.9
3	149.3	2.3	343.4	0.3	44.8
4	102.6	2.0	205.2	0.7	71.8
5	63.8	2.0	127.6	1.5	95.7
6	50.1	1.5	75.2	1.8	90.2
7	48.0	1.0	48.0	2.3	110.4
8	35.7	0.5	17.9	2.0	71.4
9	43.8	0.2	8.8	2.0	87.6
10	61.3	0.2	12.3	1.5	92.0
11	90.7	0.3	27.2	1.0	90.7
12	133.1	0.7	93.2	0.5	66.6
Mean	**93.0**	**1.2**	**126.0**	**1.2**	**74.1**

Need for further research or additional data

Further research is needed:

- to specify appropriate adjustment techniques for riverine inputs of heavy metals and pesticides;
- to explore the use of monthly data.

Recommendations

1) The ACME recommends that OSPAR INPUT provide a complete and fully documented version of the Trend-y-tector, which is not available at this stage, for evaluation.
2) The ACME recommends that OSPAR INPUT adopt some procedure for trial of the statistical methods according to the implementations recommended in the 1999 ACME report. OSPAR INPUT should be prepared for these methods to evolve to meet the particular needs of the assessment group. These needs are likely to be clarified with experience in using the package over a series of assessments. The ACME considers that there is no single all-purpose implementation (and will never be), but that for the specific purposes of trend assessment, several implementations of the methods should be developed.
3) The ACME notes that at present no single method of calculating annual adjusted loads can be recommended for all river systems/locations and all substances. However, when considering nutrients, if the input is approximately linearly related to flow/precipitation, method L1 (see Annex 2, Part 1) will perform reasonably well and the ACME recommends that it be used on a trial basis.
4) The ACME notes that at this stage it is not possible to quantify generally the gain of analysing monthly data instead of annual data for trend assessments, since this depends on the type of data and the methods applied.
5) The ACME notes that the use of robust trend detection methods depends on the purpose of the assessment, as stated in Section 6.4 of the 1999 ACME report. A robust approach is not always recommended, e.g., if information about alarming recent trends is given higher priority than making a statement unaffected by incidental outliers, then a robust approach should not be used.
6) If the assessment of the adjustment concept described in HARP Guideline 7 is critical, the ACME recommends that OSPAR INPUT specify the method to be examined.
7) Since the annual mean concentration does not necessarily reflect the effectiveness in reduction of inputs, not even in the long run, the ACME recommends that the use of the annual mean concentration should be avoided.

References

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ICES. 2000. Report of the ICES Advisory Committee on the Marine Environment, 1999. ICES Cooperative Research Report, 239: 53–65.

OSPAR. 1999. OSPAR Working Group on Nutrients and Eutrophication (NEUT). Draft Guideline 7: Quantification and Reporting of the Monitored Riverine Load

of Nitrogen and Phosphorus, including Water Flow Normalization Procedures. OSPAR Commission, London.

6.6.2 Statistical methods for designing sampling allocation strategies and assessing and analysing monitoring programmes

Request

This is part of the continuing ICES work to provide advice on the development of effective methods for designing monitoring strategies.

Source of the information presented

The 2000 report of the Working Group on Statistical Aspects of Environmental Monitoring (WGSAEM) and ACME deliberations.

Status/background information

The 2000 meeting of WGSAEM considered several statistical methods for designing sampling allocation strategies and for assessing and analysing monitoring data:

a) inspecting monthly time series with the use of Box-and-Whisker plots to reveal anomalies;

b) robust trend analysis based on a penalized LS approach;

c) Mann Kendall Test of Trend with missing observations;

d) model for the relationship between the concentration of a contaminant in an organism and the ambient concentration;

e) trend analysis for fish disease monitoring data;

f) treatment of missing values in time series of explanatory parameters.

A) Inspecting monthly time series with the use of Box-and-Whisker plots to reveal anomalies

A simple and robust method which can be used to judge monthly time series for possible outliers or interesting phenomena is based on the Box-and-Whisker plot, which can be used to identify a value as an extreme or far extreme value. For a good judgement of an extreme, four figures are presented: 1) a box plot for the values within each year; 2) a box plot for the monthly values which are corrected for the annual median values; 3) the time series with a trend which is the annual median and the median residual monthly value and an approximate 99 % confidence interval; and 4) a table with the yearly and monthly extremes which for comparative purposes are standardized by the interquartile range. By presenting the extremes with respect to year and month, a better judgement can be given concerning whether these data are some sort of outlier or reveal something that could be interesting. In Figure 6.6.2.1, an example is given for a time series of the percentage of oxygen in water for some location. In 1992, a monthly extreme for March is reported. At that time, a phytoplankton bloom was started early in the year which caused a high oxygen concentration. So avoid identifying an extreme too quickly as an outlier, because it may possibly be explained by other processes that are involved. It might be interesting also to look at the behaviour of the interquartile range, because it could be that the fluctuation of the signal can change during the years.

B) Robust trend analysis based on a penalized LS approach

A curve-fitting method was considered which combines a least-squares curve fitting with a penalty on the roughness of the curve and down-weighting extreme values. The curve fitting is done by minimizing the following penalized least-squares equation which has two penalties in it:

$$Q=\sum_{i=1}^{n} w_i^{\,2}(z_i-t_i)^2+\lambda\sum_{i=1}^{m}\left(\frac{\Delta t_i}{\Delta y_i}\right)^2+\alpha^2\sum_{i=1}^{m}(1-w_i)^2 ,$$

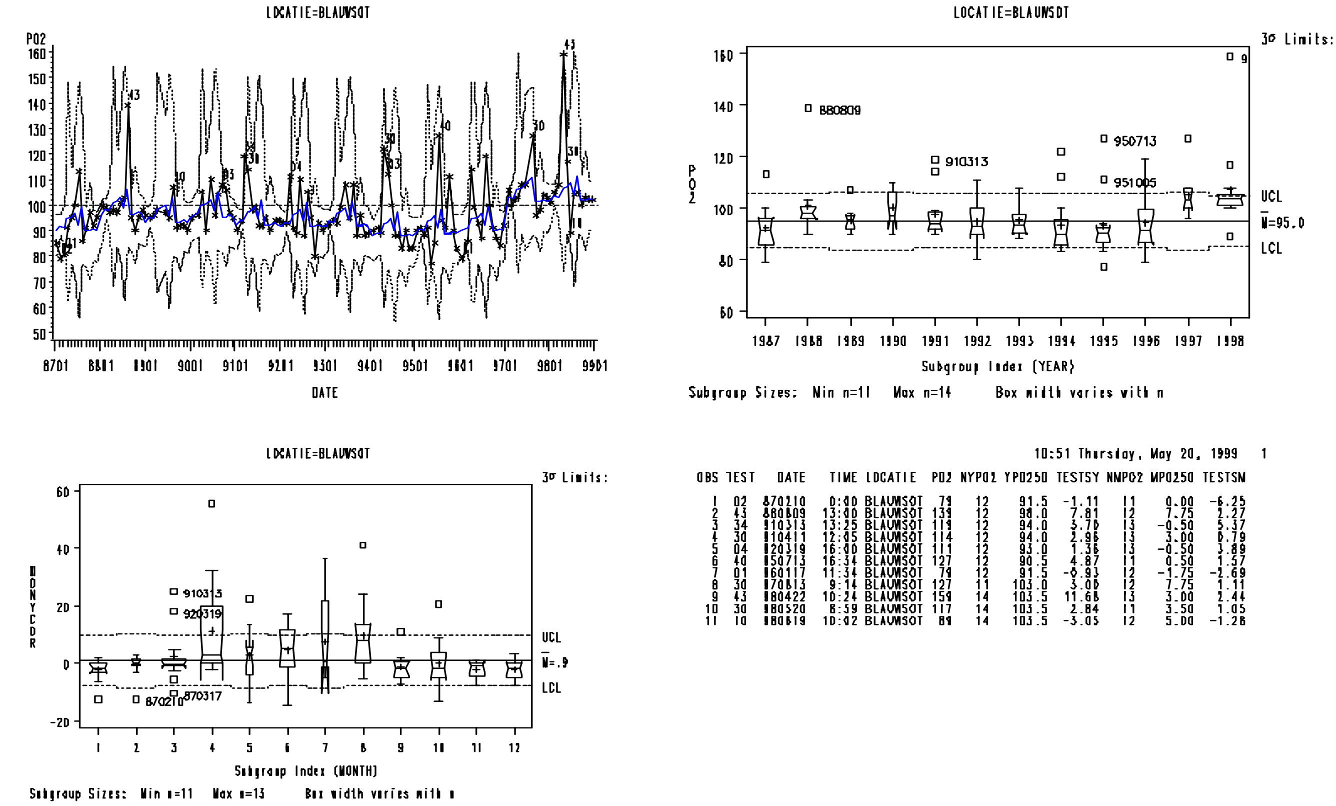
the first part is the sum of the squared differences between the measured value (z_i) and the value of the smoothed curve (t_i); the second part is the penalty on the sum of the squared distance (Δy_i) between-points weighted roughness measure of the curve (the differences between the adjacent values Δt_i); and the third part is the penalty on the sum of the squared weights (w_i.). The attempt is to give every value a weight of 1, but when α is small this endeavour is not so difficult. High λ gives a smooth curve. By introducing a weight for every value, the first part (the squared residuals) could be down-weighted for influential values. Also, this weight could be used for missing values. In that case the $w_i = 0$ and the value of z_i is interpolated. Both parameters λ and α have to be given by the user, but for the analysis of depth profiles a physical meaning can be given.

Further development of the method for the analysis of data with, e.g., seasonality looks promising. One possibility would be to include the second order differences instead of the first order differences for the roughness measure.

C) Mann Kendall Test of Trend with missing observations

One problem with the Mann Kendall test of trend is the presence of missing observations, causing uneven spacing on the time axis. A common solution is simply to ignore the missing observations and treat the time series

Figure 6.6.2.1. The template of four figures to judge a monthly time series. In the upper left corner, the original time series with the trend and the roughly 99 % confidence interval; in the upper right corner, a box plot of the annual measurements; in the lower left corner, the box plots for the median monthly value corrected for the annual median; in the lower right corner, a table with the extremes for the year and month.



10:51 Thursday, May 20, 1999 1

OBS	TEST	DATE	TIME	LOCATIE	PO2	NYPO2	YPO250	TESTSY	NMPO2	MPO250	TESTSN
1	02	870210	0:00	BLAUWSOT	79	12	91.5	-1.11	11	0.00	-6.25
2	43	880609	13:00	BLAUWSOT	139	12	98.0	7.81	12	7.75	2.27
3	34	910313	13:25	BLAUWSOT	119	12	94.0	3.70	13	-0.50	5.37
4	30	910411	12:05	BLAUWSOT	114	12	94.0	2.96	13	3.00	6.79
5	04	920319	16:00	BLAUWSOT	111	12	93.0	1.36	13	-0.50	3.89
6	40	950713	16:34	BLAUWSOT	127	12	90.5	4.87	11	0.50	1.57
7	01	960117	11:34	BLAUWSOT	79	12	91.5	-0.93	12	-1.75	-2.69
8	30	970613	9:14	BLAUWSOT	127	11	103.0	3.06	12	7.75	1.11
9	43	980422	10:24	BLAUWSOT	159	14	103.5	11.66	13	3.00	2.44
10	30	980520	8:59	BLAUWSOT	117	14	103.5	2.84	11	3.50	1.05
11	10	980619	10:02	BLAUWSOT	89	14	103.5	-3.05	12	5.00	-1.28

as if the observations were evenly spaced. However, Alvo and Cabilo (1995) described a modified version of the Mann Kendall test which incorporates the information about the spacing between the observations.

Writing $y_1, \ldots y_T$ for a series of k observations of which $T-k$ of observations 2 … $(T-1)$ are missing, the modified Mann Kendall statistic is

$$A_m = \sum_{i<j} a(i,j)$$

where

$$a(i,j) = \begin{cases} \mathrm{sgn}[r(j)-r(i)] & \text{if } \delta_i\delta_j = 1 \\ \dfrac{2r(j)}{k+1} - 1 & \delta_i = 0,\ \ \delta_j = 1 \\ 1 - \dfrac{2r(i)}{k+1} & \delta_i = 1,\ \ \delta_j = 0 \\ 0 & \text{otherwise} \end{cases}$$

where

$$\delta_i = \begin{cases} 1, & \text{if there is an observation at time } i \\ 0, & \text{if not} \end{cases}$$

and $r(i)$ is the rank of observation i scored from 1 to k.

Critical values for A_m are given in Alvo and Cabilo (1993). Approximate large sample tests using the Normal distribution can be based on the variance formulae given by Cabilo and Tilley (1999).

The powers of the modified and unmodified Mann Kendall test are compared in Cabilo and Tilley (1999), together with the corresponding versions of the Spearman rank correlation test. These powers were computed by simulation for three monotonic trend scenarios:

Scenario 1 $y_i = c_1 i + \varepsilon_i$

Scenario 2 $y_i = c_2 i^2 + \varepsilon_i$

and

Scenario 3 $y_i = c_3\sqrt{i} + \varepsilon_i$

where $\varepsilon_i \sim N(0,1)$ and the constant c is chosen so that the derivatives at observation $T/2$ are the same for the three scenarios. In Scenario 1, c_1 is obviously the slope of the (linear) trend. Figure 6.6.2.2 shows the three scenarios with $T = 10$ and $c_1 = 0.05$, together with the corresponding constant derivatives (of 0.05) at $T/2$.

The results of the simulations are extensive and somewhat complicated. As well as simulating three values of c_1, c_2 and c_3 (such that $c_1 = 0.05$, 0.1 and 0.2), for the modified Spearman and Mann Kendall tests, results were also derived for every possible arrangement of up to $T/2$ missing observations (in positions $2,\ldots,T-1$) for both $T = 10$ and $T = 12$.

When no observations were missing, the Spearman test was more powerful than the Mann Kendall test. However, with missing observations, all four methods were most powerful for some combinations of the trend, number of missing observations, and T. With increasing numbers of missing observations, the power of the modified Mann Kendall test relative to the other three methods gradually increases, until it is most frequently the most powerful.

Figure 6.6.2.3 shows the powers of both versions of the Mann Kendall test for all three scenarios with a trend of 0.05 and $T = 10$. In this case, the modified Mann Kendall test is generally more powerful.

Note that some of the variation in power is due to variation in the true size of the test around the nominal 5 %. An improvement would have been, for the purpose of the power comparisons, to have maintained a fixed true size of the test by incorporating a random element in the test as in Fryer and Nicholson (1999).

D) Model for the relationship between the concentration of a contaminant in an organism and the ambient concentration

There has been developed a simple "Superbeast" model of the relationship between the concentration of a contaminant in an organism and

- the uptake and excretion rates of the organism,
- the ambient concentration.

A similar model was presented in ICES (1993), and used to show how suitable uptake and excretion rates are important considerations in the choice of an appropriate monitoring organism. Here, the model was used to show how concentration-size relationships in the organism might evolve as a result of changes in the ambient concentration. In particular, steeper concentration-size relationships would be expected when ambient levels are decreasing. Some tentative support for the model was found in the literature.

The implications are that:

- assessment methodology for contaminant time series should be able to deal with evolving size-dependent trends;
- studying the form of size-dependent trends might help in the interpretation of changes in ambient concentration.

WGSAEM considered that the results of uptake and excretion experiments conducted, e.g., on shellfish would provide estimates of uptake and excretion rates that could be used for the further development of the Superbeast model.

E) Trend analysis for fish disease monitoring data

WGSAEM reviewed the trend analysis approach adopted by the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) during its 2000 meeting as a part of its effort to make information on the developments of fish disease prevalence accessible via the internet. Estimated prevalences are obtained from raw values by fitting a smooth curve to the latter, using a locally weighted second order polynomial regression procedure. A recent trend is defined as a significant change in the estimated prevalence during a five-year period backwards from and including the reference year (1997). A change is defined as "significant" in two different situations: (1) if the lower limit of the prevalence confidence interval for the reference year (the last year of the assessment period) is above the minimum of the upper confidence bound of the trend within the assessment period (upward trend, since there is at least one year in the last five years in which the level is significantly smaller than the prevalence in the last year), and (2) if the upper limit of the prevalence confidence interval for the reference year is below the maximum of the lower confidence bound of the trend within the assessment period (downward trend, since there is at least one year in the last five years in which the level is significantly higher than the prevalence in the last year). All other situations are labelled as "no trend".

Confidence bounds are obtained by a bootstrapping procedure, in which for each observation on the time scale a binomial sample is generated, using the actual number of fish examined and the empirical prevalence from that time point as parameters. The collection of binomial samples for all time points with actual observations constitutes one bootstrap replicate. For each replicate, a smooth trend is estimated by locally weighted regression. This procedure is repeated 1000 times, and for each estimated trend its Kolmogorov-Smirnov distance to the trend for the original observations is

Figure 6.6.2.2. The three monotonic trend scenarios, as described in the text.

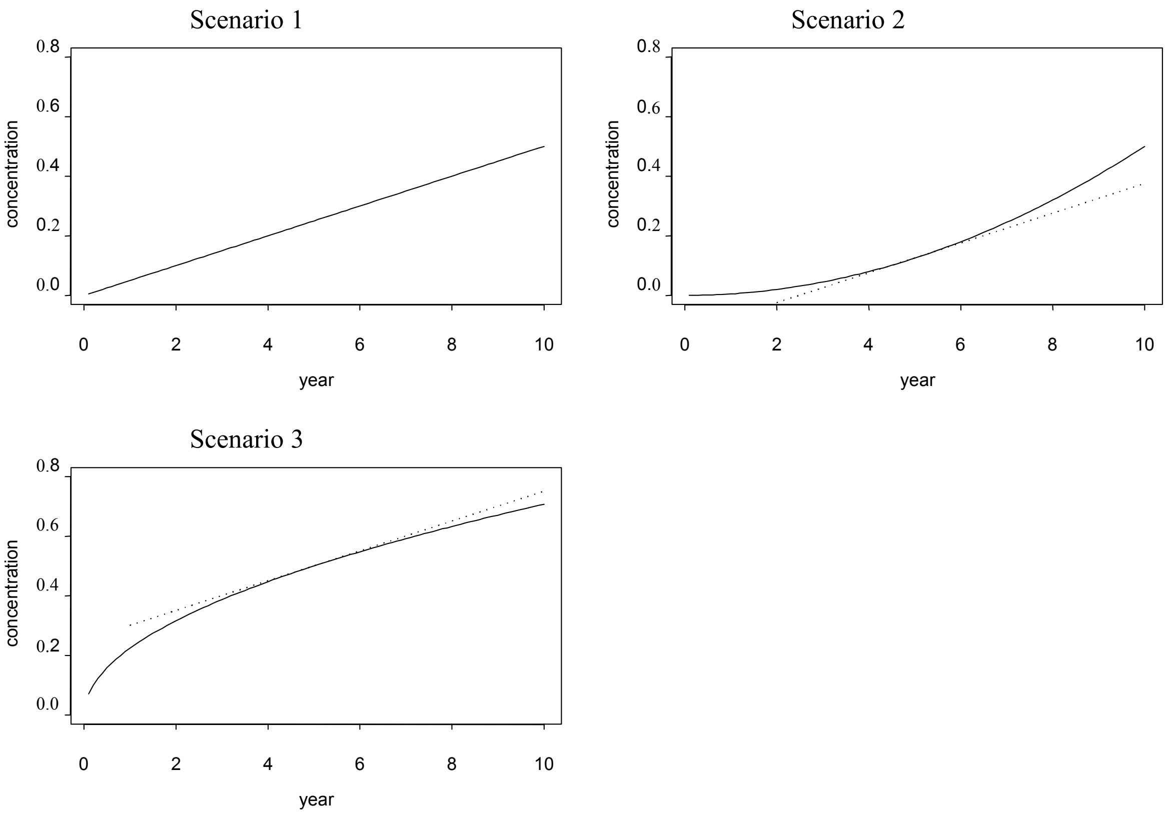
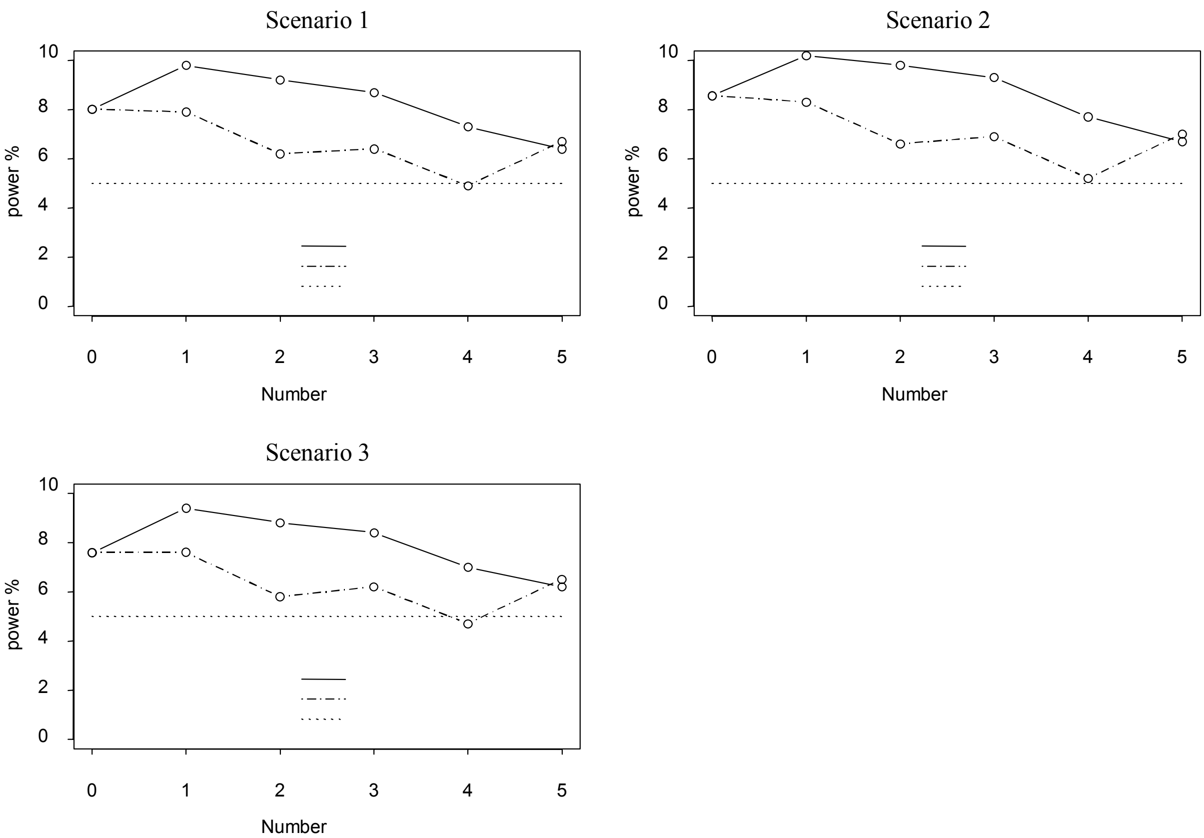


Figure 6.6.2.3. The powers of both versions of the Mann Kendall test for the three scenarios, with a trend of 0.05 and $T = 10$. The solid line is the modified Mann Kendall test, the dashed line is the Mann Kendall test, and the dotted line indicates the nominal size. The x-axis shows the number of missing observations.



calculated. Replicate trends with distances larger than the empirical 95 % quantile of these Kolmogorov-Smirnov distances lie outside the 95 % confidence bound of the estimated trend. All calculations are performed separately for observations made between October to March and for observations made between April and September, as these two periods show clearly different trends.

This approach for trend assessment is not robust against outliers in the five-year trend analysis period. The resulting danger of being erroneously alarmed by the indication of a trend induced by (an) outlier(s) is recognized, but assigned minor weight compared with the main goal that detection of a trend should initiate a more detailed analysis and possibly further action. The possible consequence of reacting on a falsely indicated trend is considered a less severe event than not reacting on a really existing trend.

F) Treatment of missing values in time series of explanatory parameters

A common way of dealing with gaps in the time series of explanatory parameters, e.g., when considering relations between fish diseases and potential explanatory parameters, is to use interpolated values to fill the gaps. The uncritical use of interpolated values in a statistical analysis, particularly simply treating them like regularly measured values, can lead to misleading conclusions. Therefore, a statistical approach was developed to incorporate interpolated values in a more appropriate way.

An interpolated value is an estimate for a missing measurement. This estimate cannot be expected to be identical with the measurement, had the latter been made, instead a random difference between both must be assumed. The distributional characteristics of this difference can be approximated by a Normal distribution with expectation zero and standard deviation s, where s is inferred from the interpolation process. With this information, the impact of using interpolated values to analyse the relation between fish disease data and a fixed set of explanatory parameters can be assessed by the following procedure:

1) Fit a smooth estimate to the time series that contains missing values.
2) Replace missing values by interpolation estimates from the smooth and record the local prediction error.
3) Generate some additional random variation for each interpolated value (Normally distributed with mean zero and standard deviation taken from the previously recorded prediction error).
4) Generate a new set of response cases, using a binomial distribution with the observed number of cases and the empirical prevalence as parameters.
5) Estimate the logistic parameters for the generated data set and record them.
6) Repeat the preceding three steps sufficiently often (1000 times gave stable results).
7) Calculate the empirical mean and quantiles of the estimated parameters and assign the corresponding p values to them.

This procedure leads to estimates for the logistic parameters, which account as well for the usual binomial variation as for the uncertainty that has been introduced by using interpolated instead of observed values. Comparing the estimates obtained with and without correction for using interpolated values led to two general observations: (i) p values for the test of the hypothesis “parameter value is zero” are increased when using the correction, which means that the hypothesis is rejected in fewer cases, and (ii) the estimated parameters move towards zero, which means that less relevance in the sense of explanatory power is assigned to the variable. Both observations are in line with what is to be expected, as the use of interpolated instead of observed values means introducing less information in the estimation process, hence the ability to detect relationships should decrease.

Need for further research or additional data

There is a continuing need for the further development of effective statistical methods, supported by an evaluation of their power and sensitivity.

References

Alvo, M., and Cabilo, P. 1993. Tables of critical values of rank tests for trend when the data is incomplete. Technical Report No. 230. Laboratory for Research in Statistics and Probability, Carleton University and University of Ottawa.

Alvo, M., and Cabilo, P. 1995. Rank correlation methods for missing data. The Canadian Journal of Statistics, 23: 345–358.

Cabilo, P., and Tilley, J. 1999. Power calculations for tests of trend with missing observations. Environmetrics, 10: 803–816.

Eilers, P.H.C., and Marx, B. 1996. Flexible smoothing with splines and penalties (with Discussion). Statistical Science, 11: 89–121.

Fryer, R.J., and Nicholson, M.D. 1999. Using smoothers for comprehensive assessments of contaminant time series in marine biota. ICES Journal of Marine Science, 56: 779–790.

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6.6.3 Spatial issues with regard to the determination of the number of samples of sediments or biota to characterize an area

Request

This is part of the continuing ICES work to provide advice on the development of effective methods for designing monitoring strategies and assessing temporal monitoring data.

Source of the information presented

The 2000 report of the Working Group on Statistical Aspects of Environmental Monitoring (WGSAEM) and ACME deliberations.

Status/background information

In 1999 the ACME had requested WGSAEM to give advice on the number of replicate samples of sediments or biota in order to characterize an area. However, a specific response on this topic would depend on the underlying objectives, e.g., estimating an area mean, mapping of contaminant concentrations, identification of hot spots, or detection of temporal changes. Further, the sampling scheme and the choice of measurement have to be considered and, ultimately, estimates of appropriate components of variability.

In order to develop the work on spatial issues, overviews of the French monitoring programmes on contaminants in sediments and contaminants in biota and the national salt water monitoring programme of the Netherlands were presented.

The French monitoring programme on contaminants in biota was initiated in 1979. Biota from eighty sites are sampled quarterly for determination of metals and organics along the French coast. Contaminants have been analysed in sediment since 1979. Each year only one area of the French coast, corresponding to 20–25 sampling sites, is investigated. Sediments are analysed for the same contaminants as for biota. Moreover, normalization parameters are measured, such as granulometry, organic carbon, carbonates, aluminium, iron, lithium, and manganese.

The national salt water monitoring programme of the Netherlands contains a chemical, a biological, and a physical part. The chemical monitoring started in 1966 and has two major objectives: temporal trend monitoring and compliance with national criteria. In 1995, an optimization and modification of the programme took place with respect to these objectives. The programme consists of chemical monitoring in water, sediments, suspended matter, and biota and biological effects (fish disease) monitoring. The JAMP monitoring programme of the Netherlands is part of this national programme. The results are published annually in a National Evaluation Report.

The biological monitoring started in 1990 and covers benthic organisms, phytoplankton, zooplankton, water birds of coast and estuaries, seabirds, and sea mammals. The main objective of this programme is to provide information on long-term developments. An evaluation of this programme is planned in 2000.

The physical monitoring deals with measurement of the bathymetry of the coastal part of the North Sea, the Wadden Sea, and the Delta: discharges, waves on the sea, water height, and water temperature. An evaluation of this programme is planned in 2001.

There is an ongoing harmonization of the three monitoring programmes.

Some preliminary statistical results were presented for contaminants in sediments in the Dutch part of the North Sea. Since 1981, every five years about 60 locations have been investigated for metals and organic contaminants. For the North Sea, four areas are distinguished which are followed over time. The sampling design consists of some fixed stations (visited every sampling time) and new “randomly” chosen stations (see Table 6.6.3.1).

Table 6.6.3.1. The number of sites which are revisited for sampling of sediments for investigation of metals and organic contaminants in the North Sea.

Number of sampling occasions	Number of sites
4	9
3	18
2	34
1	145

Assessing trends with such data can be difficult, and WGSAEM discussed various approaches that could be taken. One approach which looked promising was that described by Warren (1994), who adopted a dynamic sampling strategy in which the sampling effort each year was allocated partly to a fixed subset of sites, and partly to a new, randomly chosen group of sites. Warren also discussed the conditions under which all-fixed, all-random, and partial-replacement strategies would be superior.

Need for further research or additional data

There is a continuing need for the further development of effective sampling allocation strategies and appropriate statistical methods for assessing trends.

The ACME considered that the issue of spatial sampling design (e.g., to assess the mean level in an area) should be further developed, and encouraged further work based on case studies.

Reference

Warren, W. 1994. The potential of sampling with partial replacement for fisheries surveys. ICES Journal of Marine Science, 51: 315–324.

6.6.4 Influence of natural factors on biomarker responses in fish

Request

This is part of continuing ICES work to improve the tools available for monitoring and interpreting the biological effects of contaminants in the marine environment.

Source of the information presented

The 2000 reports of the Working Group on Biological Effects of Contaminants (WGBEC) and the Working Group on Statistical Aspects of Environmental Monitoring (WGSAEM), and ACME deliberations.

Status/background information

The ACME noted that two data sets describing biomarker distributions in fish were discussed by an informal joint session of WGBEC and WGSAEM; these emerged from the German "Stresstox" programme and from a Norwegian study.

a) German "Stresstox" data set

The German programme used dab (*Limanda limanda*) from six sites (North Sea, Baltic Sea, and one English Channel site) and involved the measurement of a suite of biomarkers (EROD, metallothionein (MT), and others) and chemical residues (conventional organochlorines, Zn, Cd, Cu, plus others) together with biological factors such as age, size, sex, condition, and reproductive status. All measurements were made at monthly intervals over a one-year period (February 1998 to January 1999), using females from a limited size range. The thrust of this programme was to attempt to establish relationships between biomarkers and the presence of both contaminants and natural ecophysiological factors. All the biomarkers studied (EROD, MT, apoptosis, etc.) showed strong seasonal variation, but seasonal maxima and minima were not obviously correlated, that is, biomarkers followed different seasonal cycles, some of which could be related to reproductive cycles (Table 6.6.4.1). Organochlorine residue concentrations and metal distributions also followed seasonal cycles, but they differed from those of the biomarkers with which they were expected to be correlated.

One objective of the OSPAR Joint Assessment and Monitoring Programme (JAMP) is the measurement of biological effects in fish once a year outside the spawning season, preferably simultaneously with measurements of relevant contaminants. This avoidance by JAMP of the spawning season was intended to avoid the possibly confounding influence of elevated steroid hormone titres on such biomarker measurements as EROD induction. The present investigation suggests, however, that maxima in annual cycles exist beyond those that occur during the spawning season. If it is required that measurements should be made out of the spawning season during periods of low biomarker values, then a suitable period for each single biomarker can be taken from Table 6.6.4.1. Analogously, in order to use a battery of biomarkers simultaneously, a period containing only marker minima may be required.

However, Table 6.6.4.1 shows that a common "low" period for, e.g., EROD, MT, and heat shock protein (HSP) does not exist. The cycle of these variables can be assumed to change to some extent from year to year. Anthropogenic influences would induce additional variation in biomarker values. In order to derive a proper assessment of biomarkers, particularly to identify anthropogenic effects, the natural annual cyclical variation of a biomarker must be accounted for.

It should be noted that these conclusions about the existence of annual cycles in biomarker values are based upon the observation of a one-year cycle at a single location, and should therefore not be taken on their own as a sufficient reason to modify monitoring policy. It is assumed, however, that such cycles may exist more generally, although the shapes of the cycles actually found cannot simply be carried over to other times or locations. A confirmation of this assumption requires the analysis of additional data, including information on varying residue concentrations that may contribute to the observed seasonality in biomarker responses. However, the working hypothesis is that the majority of the seasonality is natural, and it is concluded that the observed annual cycles should be a major factor in the interpretation and evaluation of biomarkers.

b) Norwegian data set

The Norwegian programme focused on 300 cod (*Gadus morhua*) from six stations in some industrialized fjords and at reference sites, and involved sampling males and females once annually over a 2–3 year period (1996–1998). Again, a suite of biomarkers and chemical residues was examined along with "natural" variables. Some biomarkers were correlated with expected chemical causes (e.g., δ-aminolevulinic acid dehydratase (ALA-D) and Pb), but correlations between others (e.g., EROD and organochlorines (OCs), or MT and metals) were less clear. Statistical analyses of this data set were less complete than in the German study, and it would be premature to draw many conclusions yet.

Table 6.6.4.1. Maxima (dark shading) in annual cycles of biomarkers and related parameters measured in the liver of dab from the North Sea (February 1998 to January 1999). Data on water temperature were made available by the Federal Maritime and Hydrographic Agency of Germany. Lipid = lipid content of liver; MT = metallothionein; SSF = DNA strand breaks; HSP = heat shock protein; GSI = gonadosomatic index; EROD = ethoxyresorufin-*O*-deethylase.

	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan
Spawning	□	□	□	□								
Temperature												
Lipid												
Zinc												
MT-I												
MT-II												
Apoptosis												
SSF												
HSP												
GSI*												
EROD**												

*Saborowski, R. 1996. Zur Ökophysiologie der Kliesche, *Limanda limanda* (L.). Einfluß saisonaler Zyklen auf das hepatische Entgiftungssystem. Ph.D. Thesis, University of Hamburg, Department of Biology, 168 pp. **Vobach, M., and Kellermann, H.J. 1999. Entgiftungsstoffwechsel der Kliesche (*Limanda limanda*). *In* Jahresberichte 1998. Bundesforschungsanstalt für Fischerei, Hamburg.

However, the biomarkers were analysed with GLM models for log-transformed variables, with and without station and/or year effects included. The main results of this analysis are as follows:

Metallothionein (MT): There was a clearly significant negative relation between MT and zinc, with MT being proportional to the inverse square root of zinc concentrations, both with and without station*year included. However, Zn accounts for at most 15 % of the total variance of MT over the stations with data. There also appears to be a relation to fat content of the liver. For Cu and Cd in liver and Hg in muscle, there are less clear relations with MT, with significance depending on what other terms are included in the model.

EROD: Without separation by station, EROD is significantly correlated to the contaminants OH-pyrene (in bile), Cd, p,p′-DDE and Hg (in muscle), and also to physiology, weight, and liver fat condition measures. With station included as a factor, only Cd remains significantly correlated to EROD; for the other contaminants, the correlation is mainly due to covariance of differences between stations with station differences of EROD.

ALA-D: Without station included as a factor, there is a significant positive correlation with Zn, and a negative correlation with Cd and Hg. Cadmium is the most important of these three contaminants, accounting for about 25 % of the total variance. When station is included as a factor, the relations are weaker, but still significant and in the same direction. Lead was not included in the analyses due to a large number of values less than the detection limit.

Several general points emerge from the German and Norwegian studies:

1) The data illustrate the need to consider natural seasonal cycles as factors which could affect either "baseline" biomarker concentrations or activity, but also the sensitivity of the biomarker response. In the case of EROD, for example, "baseline" activity may vary about five-fold over a seasonal cycle, depending on species (see also, e.g., Edwards *et al*., 1988), and at least in trout, the inducibility of EROD varies with reproductive maturity (Stegeman and Chevion, 1980). Therefore, in making comparisons between sites (over space or time) such potentially confounding variables must be accounted for. (It is worth noting, in passing, that such variables should also be corrected for when comparing chemical residue distribution and/or trends in fish.)

2) Biomarker measurements should not be applied unthinkingly, or even as a general "screening" approach, because of the difficulty of interpreting data in the absence of detailed knowledge of the role of confounding factors (such as the natural factors listed above, or hitherto unexpected factors), or in the absence of dose-response "calibration" data. Instead, the application of biomarker measurements should be hypothesis-driven. Given the uncertainty about how to interpret *absolute* biomarker measurements at present, biomarkers are best applied in a comparative setting, where spatial comparisons in biomarker response are made between putatively impacted and "reference" sites, or along a demonstrated contamination gradient.

3) Biomarker data are perhaps best used at present to complement chemical residue data in a total "weight of evidence" approach to detecting and recording environmental trends over space or time.

4) These conclusions raise some strategic questions for marine monitoring organizations. For example, if they wish to use biomarker measurements as a screening approach to "lead" chemical analyses, the biomarker response to specific chemical causes will usually have to be carefully calibrated (i.e., dose-response relationships established) and the significance of confounding factors assessed. It will be expensive to do this for more than a few potential monitoring species; it may therefore be necessary to try to identify a widely distributed species for which it would make sense to invest the effort to carry out such calibrations. If it is decided not to refine biomarker measurements in this sense, should biomarker measurements simply remain as an interesting complement to chemical residue measurements? It is probable that some compromise between these two extremes will emerge.

Need for further research or additional data

The issue of annual cycles in biomarkers should receive further attention, including the collection of more data. Furthermore, when assessing annual cycles of biomarkers, the potential effect of similar cycles in contaminants should be studied. Finally, when developing new biomarkers, it is vital to establish clear dose-response relationships for relevant contaminants in species of interest. It is important that this research is carried out, because failure to understand the natural causes of variability in biomarkers, and the dynamics of dose-response relationships of contaminants, could lead to misinterpretation of pollution effects.

Recommendation

ICES recommends that, at the present state of knowledge, biomarkers are best used in a "weight of evidence" approach together with other biological and chemical measures of pollution. Great care should be taken to distinguish between the effects of natural and anthropogenic influences on biomarkers.

References

Edwards, A.J., Addison, R.F., Willis, D.E., and Renton, K.W. 1988. Seasonal variation of hepatic mixed function oxidases in winter flounder (*Pseudopleuronectes americanus*). Marine Environmental Research, 26: 299–309.

Stegeman, J.J., and Chevion, M. 1980. Sex differences in cytochrome P-450 and mixed-function oxygenase activity in gonadally mature trout. Biochemical Pharmacology, 29: 553–558.

6.6.5 Implications of the outcome of the VIC Programme on the monitoring of temporal trends in contaminants in fish

Request

This is an informal request by the OSPAR Working Group on Concentrations, Trends and Effects of Substances in the Marine Environment (SIME) for ICES to review the results of this programme that has been conducted under the framework of OSPAR.

Source of the information presented

The 2000 report of the Working Group on Statistical Aspects of Environmental Monitoring (WGSAEM) and ACME deliberations.

Status/background information

The Voluntary International Contaminant (VIC) Programme on the monitoring of temporal trends in contaminants in fish was designed to provide information about small-scale temporal and spatial variations, essential for providing estimates of components of variance that may improve the efficiency of the OSPAR monitoring programme. The VIC Programme was conducted within the framework of OSPAR, but review of the outcome by ICES was informally requested.

The information presented to WGSAEM was based on a paper previously presented to the 2000 meeting of OSPAR SIME, in which the data submitted by the Netherlands and Norway to the VIC Programme had been analysed. From the Netherlands, data for flounder were submitted, collected at different sites and at two times each year from the estuary of Westerschelde over the years 1996–1998, with both individual and pooled samples. From Norway, there were data for cod and flounder collected at different sites and/or times in the Oslofjord and in Sørfjord and Hardangerfjord on the west coast, with subsets covering both temporal and spatial small-scale variations. From Sweden there were data for herring from different locations in the southwestern part of the Baltic Sea, the Kattegat, and the Skagerrak. The within-year variation in these data is mainly spatial, and describes variation over larger distances in open waters.

The results of the analysis so far indicate that, at least for some species and contaminants, the within-year variance could be substantially reduced by sampling from more than one site or repeating the sampling over time. Depending on cost components, this might result in improved efficiency per cost.

To demonstrate how these data might be used, Table 6.6.5.1 shows the estimated components of between-site, between-time, and between-individual fish variances for log CB153 measured in cod livers from the Oslofjord. The spatial scale is 5–10 km within the total area of 20 km × 50 km of the Oslofjord. The temporal scale spans two weeks.

Table 6.6.5.1. Estimates of variance components for CB153 in cod liver derived from ANOVA of the VIC data for station 30B (location Maasene) in the inner Oslofjord.

	CB153_LI
σ_t^2 between *Times* within year	0.114
σ_s^2 between *Sites* within year	0.125
σ_I^2 between *Individuals* within year	0.256
Total variance for individual fish within year	0.495

Table 6.6.5.2 provides an estimate of the between-year variance obtained using the full JAMP data for two stations (15B and 23B) with relatively small yearly fluctuations.

Table 6.6.5.2. Estimated between-year component of variance for CB153 in cod liver using the full JAMP data for two stations (15B and 23B) with relatively small yearly fluctuations.

	CB153_LI
Total between-year variance (20–25 fish samples)	0.055
Within-year variance for individual fish: 0.4–0.5	0.47
Within-year variance for mean over 22 fish	0.021
σ_y^2 between-year variance component	0.034

These estimates can be combined to give an estimate of the total between-year variance of an estimated annual mean log concentration, and evaluate the corresponding power, e.g., to detect a specific linear trend in a given number of years (cf. Nicholson *et al.,* 1997). We can then explore the consequences of different allocations of sampling effort within a year.

Figure 6.6.5.1 gives plots of the powers to detect trends of 5 % and 10 % per year after ten years, assuming a 5 % significance level and 24 fish sampled per year. Fish are assumed to be collected in equal numbers from each site visited on each sampling occasion such that the total number of fish is constant. The power is plotted against the number of sites.

If spatial and temporal variations are generated by random fish movements, they may be measuring the same thing. In this case, only one type of variation should be included, or they could be pooled. The solid lines in each figure therefore show the change in power assuming that the between-time variance is zero, with the 24 fish sampled at 1 site, 2 sites, and so on, to each fish coming from a different site. If variations in space or with time are the same, the curves would apply to the number of sub-samples on the horizontal axis, irrespective of whether there is a difference in time or location between sub-samples.

The dotted lines assume that both between-site and between-time variance components are present. From bottom to top, the lines correspond to sampling at one time from up to 24 sites, 2 times from up to 12 sites each time, and so on up to 12 times. Together, the points on the dotted lines represent all combinations of site number *S* and number of times *T* such that $S \times T = 24$.

The power corresponding to the current sampling guidelines of 24 (actually 25) fish sampled on one occasion from a single site falls somewhere between the point on the lowest dotted line at sites = 1 (and times = 1) and the solid line at sites = 1. Clearly, for this example there is considerable opportunity for improvement, depending on the variance between sites and times.

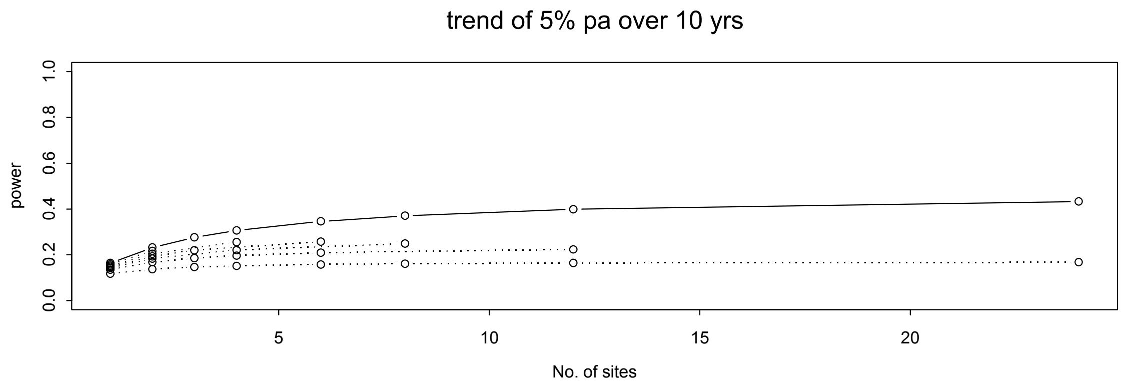
This means that Figure 6.6.5.1 may be used to allocate 24 analytical samples in an optimized way in terms of the detectability of a linear temporal trend. Assuming that the migration area of the fish is small, both the spatial and the temporal component of the estimated variance have to be taken into account and the circles of the dotted lines represent the corresponding power. Maximum power can be achieved either with six times and four sites or with four times and six sites. In both cases, the resulting power for a trend of 10 % per year over ten years is about 73 % (see lower figure). If the contract with the fisher restricts the number of sampling times to two, best results will be obtained with twelve sites sampled and the power for a 10 % per year trend will be about 63 %.

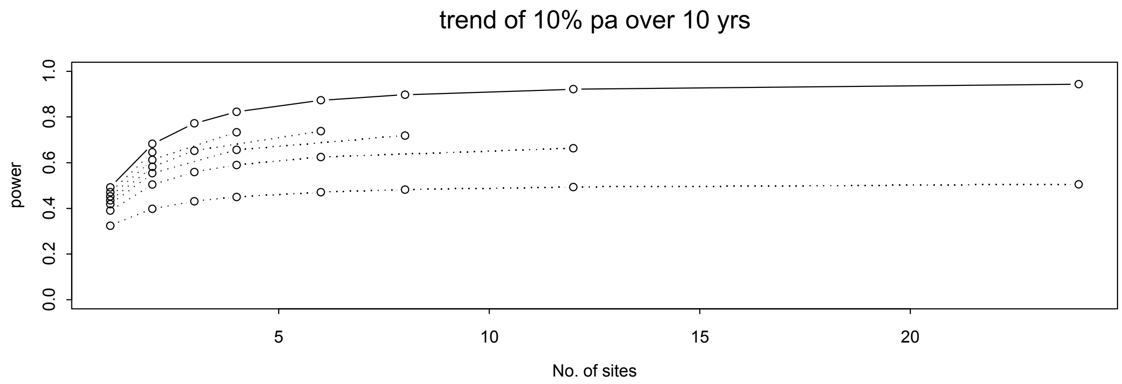
In case that the migration of the fish is rapid, increasing the number of sampling times will not be relevant for improving the power. Then a maximum power of approximately 90 % for a trend of 10 % per year over ten years can be achieved by increasing the number of sites to 24.

The example given above highlights the importance of taking the migration behaviour of a certain fish species in a certain area into account when designing a sampling programme for trend monitoring and, hence, that this is best done in a close cooperation between biologists, statisticians, and chemists.

WGSAEM noted that the data sets available for these kinds of analysis are very patchy with respect to species, geographical and temporal scales, and sampling design. Thus, the results are fragmentary and, moreover, are based on low degrees of freedom. WGSAEM felt that it would be worthwhile to extend the database by locating and incorporating other similar data sets.

Figure 6.6.5.1. The power to detect a trend of 5 % and 10 %, respectively, per year (pa) over ten years as a function of sampling strategy (number of occasions and sites).





Need for further research or additional data

In order to be able to investigate the general applicability of the results regarding other species and contaminants, more data are needed.

Accordingly, the ACME recommends that WGSAEM specify which types of data are needed to improve the model.

Reference

Nicholson, M.D., Fryer, R.J., and Ross, C. 1997. Designing monitoring programmes for detecting trends in contaminants in fish and shellfish. Marine Pollution Bulletin, 34: 821–826.

6.7 Evaluation of the Effectiveness of Monitoring Programmes in Determining Trends against a Background of Natural Fluctuations

Request

This is part of the continuing work by ICES on issues related to monitoring the marine environment.

Source of the information presented

The 2000 reports of the Working Group on Shelf Seas Oceanography (WGSSO), the Marine Chemistry Working Group (MCWG), and ACME deliberations.

Status/background information

The ACME noted that, at the MCWG meeting in 1999, it was concluded that no unique, simple approach for monitoring nutrient fluxes from estuarine environments seems possible and that it would seem unrealistic to build a general strategy for monitoring nutrient inputs to the coastal waters. Ecological modelling is seen as a possibility for optimizing the existing heavy monitoring programmes.

It is recognized that there can be essentially two different kinds of strategic aims for a monitoring programme: to detect trends against a background of natural variations and/or to classify the environmental state by using prescribed classes (e.g., contaminated or not contaminated) (ICES, 1999, 2000). Which strategy one chooses depends on existing knowledge, secondary aims, etc.

The usefulness of the present monitoring strategy as well as of the traditional cruise-based monitoring has been questioned. It was pointed out that models are useful for short-term forecasting and for describing scenarios. The more traditional monitoring has proved to be useful for detecting, e.g., long-term changes in the environment. It was also noted that since cruise-based monitoring has a long tradition, it might sometimes be difficult for chemists to see what kind of possibilities modelling can offer and how and what kind of data could be used for modelling work. Marine chemists are not used to this kind of immediate application of their data. Modelling was considered to offer a good capability to be used, e.g., for early warning systems for plankton blooms and bathing water quality.

The modellers' request for greater volumes of less accurate data was discussed at length. MCWG concluded that it is still important to produce quality assured data, but the purpose of the data determines the quality level. In any case, the data user has the responsibility of selecting data that fulfill his/her quality requirements. This requires that the quality of the data is known. To fulfill the wish of modellers to have greater volumes of data with high speed, O_2, CO_2, fluorescence, and pH were identified as possible parameters where continuous measurements could be started. The importance of having correct boundary values in the models was stressed, since in some cases only a small difference in the value used might cause a major difference in the final model result. Attention should always be drawn to the interpretation of the model results and possible misuse of the results.

The ACME was informed that WGSSO discussed the difficulties in assessing the quality of nutrient monitoring programmes. The statistical approaches vary widely. WGSSO has previously studied several monitoring programmes, as well as reviews from, e.g., Canada, Germany, Norway, and Sweden. The Working Group agreed on the following:

- monitoring programmes should focus on fluxes (rather than levels), sources and sinks, and on estimating the variability in frequencies and scales;
- monitoring programmes should, increasingly, use models as a support when designing programmes, producing results, and interpreting the findings;
- monitoring programmes should, in general, have an appropriate balance between highly frequent observations and high spatial resolution.

When designing a monitoring programme, independent of the strategy chosen, it is necessary to:

- specify the aim (identify relevant environmental threats, for example, eutrophication);
- define requirements (how small a trend should the programme be able to detect);
- identify relevant variables (nutrient content, type of annual value, etc.) from knowledge about the general conditions of the area of concern (temporal and spatial variability, etc.);
- evaluate the usefulness of these variables (detectable trend) in relation to the expected change;
- specify requirements and strategy (power, correlation and use of possible co-variables (Kendall and Ord, 1990; Hirsch and Slack, 1984)).

WGSSO also recognized the possible use of so-called fuzzy logic in developing management tools.

To facilitate the preparations and planning for a possible joint meeting or workshop, the two Working Groups concluded that it is important to:

- continue the communication between modellers and chemists;
- define clearly the problems for which models are needed and define clearly the data needs in relation to time and space as well as to the quality of data;
- start discussions on restructuring monitoring programmes and reallocating existing resources used for monitoring and modelling;
- continue discussions on future cooperation on modelling, not only among physical modellers, but trying also to include chemical and biological modellers who are devoted to process modelling.

The ACME recognizes that it is a complex problem to design a general monitoring programme applicable for all areas and chemical compounds (nutrients, contaminants). Therefore, the initial focus should only be on nutrient monitoring. A workshop concerned with the design of monitoring programmes to determine nutrient trends against a background of natural fluctuations should be held, preferably in autumn 2001 and possibly in cooperation with other relevant organizations. Participants should include experts on monitoring, modelling, and statistics.

References

Hirch, R.M., and Slack, J.R. 1984. A nonparametric trend test for seasonal data with serial dependence. Water Resources Research, 20(6): 727–732.

ICES. 1999. Report of the ICES Advisory Committee on the Marine Environment, 1998. ICES Cooperative Research Report, 233: 23–25, 34–36.

ICES. 2000. Report of the ICES Advisory Committee on the Marine Environment, 1999. ICES Cooperative Research Report, 239: 40–41.

Kendall, Sir M., and Ord, J.K. 1990. Time series (3rd Ed.). Edward Arnold, London.

6.8 Biological Community Studies

6.8.1 Guidelines for studies of epibiota

Request

This is part of continuing ICES work on the development of guidelines for marine biological monitoring.

Source of the information presented

The 2000 report of the Benthos Ecology Working Group (BEWG) and ACME deliberations.

Status/background information

At the 2000 BEWG meeting, a sub-group was formed to take forward the work on the preparation of guidelines for sampling and objective community description of the epibiota of soft sediments and hard-bottom sub-strata, including QA matters. It was agreed that the goal should be the production of a report, aimed at a wide readership, for publication in the *ICES Techniques in Marine Environmental Sciences* (TIMES) series. It was also agreed that, in order to maintain continuity with earlier TIMES reports dealing with benthos sampling methods and to ensure that the output is of manageable size, the subject matter will be confined to the subtidal zone. Progress was made in further developing a structure for these guidelines and in generating an initial text. Further work will be conducted intersessionally, including contact with possible contributors who were not in attendance at the BEWG meeting, so that a more complete draft can be submitted to the 2001 BEWG meeting for review. The contents of the guidelines will have the following headings:

1) Introduction

   This will contain the background and scope of the report; the definition, role, and importance of the epibiota; objectives of epibiota studies.

2) Design and Conduct of Epibiota Surveys

   This will cover sampling design; navigation, nature and limitations of sampling gear; parallel environmental measures.

3) Guidelines on Sampling Methodology

   The contents will include destructive sampling; towed gear, trawls, dredges, grabs/cores; suction samplers, diver-operated, sediment profile imagery; non-destructive sampling; acoustics, video, photography, direct visual, platforms, drop-frame, tripod, diver, towed bodies, ROV/AUV; manned submersibles.

4) Sample Processing

   This will cover procedures for use in the field; laboratory; still/video images; biological samples.

5) Approaches to Describing Assemblage Types

6) Quality Assurance of Epibiota Studies

7) References

The ACME felt that these guidelines, when completed, will be very useful for cooperative studies of epibiota in the ICES area.

6.8.2 Progress in understanding the dynamics of harmful algal blooms

Request

This is part of continuing ICES work to support research and collect information on this issue, owing to the health and economic problems associated with the worldwide occurrence of harmful and/or toxic phytoplankton blooms.

Source of the information presented

The 2000 report of the ICES/IOC Working Group on Harmful Algal Bloom Dynamics (WGHABD) and ACME deliberations.

Status/background information

WGHABD provided a retrospective analysis of the work conducted during its ten-year existence. The different subjects treated during this period have had varied degrees of success.

- The design of pilot studies has been useful in that they have initiated discussions between physicists and ecologists. However, since ICES cannot facilitate the funding of scientific collaboration, the pilot studies did not reach the implementation level.
- The WGHABD report highlights the need for detailed planning prior to the conduct of any Workshop.
- A fruitful dialogue has been initiated between physicists and ecologists.

The ACME noted that a major outcome of WGHABD activity has been the establishment of a dialogue between physicists and biologists: it is recognized by the community that this dialogue was essential to the establishment of GEOHAB—the SCOR-IOC Programme on Global Ecology and Oceanography of Harmful Algal Blooms (see Annex 3). The ICES support to GEOHAB through close interactions between WGHABD and the Working Group on Shelf Seas Oceanography (WGSSO) should be continued.

WGHABD has pursued efforts in defining the HAEDAT (Harmful Algal Event database) reporting format and procedures. This database will be operated by the IOC Vigo Communication Center and, being worldwide, will be a valuable tool. The establishment of this database was made possible through a sustained effort over many years.

Two new items were discussed by WGHABD. The use of fossil records was described by an invited specialist. It was noted that these records will be needed when examining causes in relation to recent time series.

To date, benthic dinoflagellates have not been considered as causative of harmful algal events outside the tropical regions. However, a case was presented from Canada in which *Prorocentrum lima* was the DSP causative organism. This new finding, if verified in other ICES countries, implies a need for changes in monitoring procedures and will require a completely new approach for modelling in order to understand toxin accumulation.

During the joint meeting of WGHABD with WGSSO, different parameterizations in models of phytoplankton population dynamics were presented and discussed in terms of the implications for the complexity required in physics. It was recognized that significant progress in forecasting harmful algal events will require real-time observation systems and data assimilation procedures.

Having reviewed the above material, the ACME recommended that ICES support the development and implementation of GEOHAB in collaboration with IOC and SCOR.

6.8.3 Zooplankton ecology issues

Request

This is part of continuing ICES work on zooplankton issues to improve and standardize methods for marine biological monitoring.

Source of the information presented

The 2000 report of the Working Group on Zooplankton Ecology (WGZE), the preface to the *ICES Zooplankton Methodology Manual*, and ACME deliberations.

Status/background information

Zooplankton taxonomic skills

The ACME noted the concern of the WGZE about the loss of taxonomic expertise within the ICES zooplankton community, expressed in the WGZE reports in 1998 and 1999. The WGZE has underlined the importance of the use of taxonomy in zooplankton ecology studies. There is a general trend of more emphasis on behaviour and population dynamics in ecological studies, e.g., in GLOBEC programmes. There is also the issue of biodiversity in pelagic ecosystems. Therefore, there is a very strong case not only to preserve but also to improve zooplankton taxonomic skills within the ICES community. This is also a contribution to the quality assurance work on zooplankton species identification (see Harris *et al*., 2000).

As one activity to propose future collective actions in this field, a Workshop on the Taxonomy of Calanoid Copepods was hosted by the TerraMare Research Institute (Wilhelmshaven, Germany) from 14–17 May 2000. The objectives for the workshop were to:

- improve and intercalibrate the present taxonomic knowledge among scientists;
- recommend, strengthen, and initiate future taxonomic research;
- review existing identification keys for the North Atlantic area of ICES.

The workshop focused on the Calanoida, dealing with genetic taxonomy advances and the application of biochemical methods. Practical taxonomic work on specific taxa as well as discussions on difficulties in determination, new systematic, recommended keys, standards for archiving data and taxonomic information, and production of regional checklists for pelagic copepods were also on the agenda.

Additional comments

The ACME shared the concern of WGZE regarding the loss of expertise within the ICES zooplankton community and encourages ICES Member Countries to provide experts for future work on these issues.

ICES Zooplankton Methodology Manual

The ACME noted that much of the work of ICES is carried out by its many Working Groups, and the reports of the activities of the Working Groups provide a major input to the Annual Science Conference each year. On occasion, Working Groups produce other scientific outputs, and a recent example is the *ICES Zooplankton Methodology Manual,* a compilation of 684 pages which has just been published by Academic Press, as a result of the work of the Working Group on Zooplankton Ecology (WGZE) over a number of years.

Zooplankton are the diverse, delicate, and often very beautiful, assemblage of animals that drift the waters of the world's oceans. These microscopic organisms play a key role in the pelagic food web by controlling phytoplankton production and shaping pelagic ecosystems. In addition, because of their critical role as a food source for larval and juvenile fish, the dynamics of zooplankton populations, their reproductive cycles, growth, reproduction, and survival rates are all important factors influencing recruitment to fish stocks. It is this latter role which has made zooplankton ecology of particular interest to ICES.

In 1992, ICES established a Study Group on Zooplankton Production, which decided at its first meeting to produce a Zooplankton Methodology Manual, recognizing the need for improvements and standardization in methods for studying this important and challenging group of organisms. To assist in the review of methods and to provide input to the issue of standardization and improvement of methods, three special workshops were convened, covering zooplankton sampling methods and production methods. Results from these workshops, as well as extensive other work, have been utilized by the successor of the Study Group, the Working Group on Zooplankton Ecology (WGZE), in its task of completing work on the Manual.

The scope of the Zooplankton Methodology Manual is to provide an updated review of basic methodology used in studies of zooplankton, including recommendations on improvements, harmonization, and standardization of methods. The chapters aim to maintain a balance between being introductory and comprehensive. They provide an overview of methods that are useful, for example, to graduate students who are starting in a new field. They emphasize the sources of error and the strengths and weaknesses of methods for various purposes and tasks. It was not possible to go into detail for all methods, and reference to recent reviews and detailed descriptions of methods was used where appropriate.

Each chapter begins with a review of methods, which in most cases is accompanied by recommendations regarding the choice and conduct of methods. These reviews consider the background and history of the methodology, the basic principles, sources of variability, equipment and procedures, comparative evaluation of alternative methods, general recommendations, and extensive literature references. Where possible, detailed descriptions of standard protocols have been included. The aim was to give practical instructions on how to carry out particular measurements and procedures. Equipment, procedures, data analysis and interpretation are described, where possible. These protocols either define standard methods, or give examples of little-known methods. If many methods or many instruments are used, guidance was given on the most highly recommended or the most often used. In some cases, it proved difficult to propose an agreed standard protocol. It was, however, possible to provide guidelines that reduce the variability in methods and contribute towards harmonization and standardization.

The various chapters of the Manual have been reviewed by the WGZE, and in addition peer reviewers from outside this group have evaluated each chapter independently. The ACME expressed gratitude to these reviewers for their valuable contribution to the overall project.

The ACME noted that WGZE has encouraged and coordinated zooplankton monitoring activities in the ICES area, and this Manual should contribute to these activities. Similarly, the development of major international initiatives with a particular focus on zooplankton, particularly the IGBP/SCOR/IOC Global Ocean Ecosystem Dynamics (GLOBEC) project and the Living Marine Resources module of the Global Ocean Observing System (GOOS-LMR), make the publication of this Manual particularly timely. While not formally adopted by either programme, the ICES Zooplankton Methodology Manual will contribute significantly to the standardization of methodology that both GLOBEC and GOOS-LMR strongly endorse. The 2000 meeting of the WGZE in Hawaii, with guests from the PICES zooplankton community, is a further step in the process of zooplankton methods standardization between ocean basins.

The ACME further took note that preparation of the Zooplankton Methodology Manual has been a team effort. The members of the WGZE and the Editors have led in this, over the years of development. The ACME expressed its thanks to Dr R. Harris, to the editorial board, and to all the contributors to the ICES Zooplankton Methodology Manual.

Reference

Harris, R.P., Wiebe, P.H., Lenz, J., Skjoldal, H.R., and Huntley, M. (Eds.) 2000. ICES zooplankton methodology manual. Academic Press, London, UK. 684 pp.

7 QUALITY ASSURANCE PROCEDURES AND INTERCOMPARISON EXERCISES

7.1 Quality Assurance of Biological Measurements in the Baltic Sea

Request

Item 1 of the 2000 requests from the Helsinki Commission: to coordinate quality assurance activities on biological and chemical measurements in the Baltic marine area and report routinely on planned and ongoing ICES intercomparison exercises, and to provide a full report on the results.

Source of the information presented

The 2000 report of the ICES/HELCOM Steering Group on Quality Assurance of Biological Measurements in the Baltic Sea (SGQAB) and ACME deliberations.

Status/background information

The development of quality assurance guidelines and procedures for the biological measurements in the HELCOM Cooperative Monitoring in the Baltic Marine Environment (COMBINE) Programme has been carried out by the ICES/HELCOM Steering Group on Quality Assurance of Biological Measurements in the Baltic Sea (SGQAB) since its establishment in 1992. The progress made since the 1999 meeting of ACME was reviewed, as described below.

SGQAB has continued the review of Part B "General Guidelines on Quality Assurance for Monitoring in the Baltic Sea" of the COMBINE Manual concerning their applicability for biological analysis and data.

In order to assess the present and near-future use of the new primary production method ("Working manual and supporting papers on the use of a standardized incubator technique in primary production measurements" by F. Colijn and L. Edler) in the HELCOM area, SGQAB developed a questionnaire to be distributed to the laboratories. The results will be reviewed by SGQAB at its 2001 meeting.

Concerning the future specific QA aspects of phytobenthos monitoring, SGQAB decided to request Finland to gather information on experience in the use of the new phytobenthos monitoring guidelines in the COMBINE Manual. The 2001 meeting of SGQAB will consider the results.

SGQAB reviewed the work of the taxonomic training courses on phytoplankton and benthos. The Phytoplankton Workshop in October 1999 concentrated on the taxonomy of blue-green algae and the diatom genus *Chaetoceros*. The draft ICES reporting formats for biological data, the HELCOM phytoplankton data entry program, and the current status of the Baltic Phytoplankton Checklist were reviewed. The Macrozoobenthos Training Course, held in February 2000 as a joint HELCOM and BEQUALM workshop, concentrated on identification problems of certain groups: Amphipoda, Isopoda, Decapoda, Oligochaeta, and Acari. A summary of sampling techniques in use, with emphasis on new and special methods for problematic habitats, was given.

The current status of the manual for chlorophyll *a* measurements, produced by the Marine Chemistry Working Group (MCWG) and the Working Group on Phytoplankton Ecology (WGPE), was reviewed. In the manual, acetone is recommended for use as the extraction solvent. In the COMBINE Manual, ethanol is used as the solvent. An assessment of the outcome of exercises to compare the results using these two solvents should be carried out.

Quality criteria for biological data were considered in accordance with the questions to be answered in the assessments. For biological data, an outline of the quality criteria is not unambiguous: the species data can be compiled using various taxonomic levels (e.g., species, genus, family) with different QA requirements. The quality criteria for data are very different when assessing, e.g., the presence or absence of potentially toxic species or long-term trends in biomass values. In order to use species data to analyse long-term changes, an adapted QC of taxonomic information is essential. The controlled updating of the taxonomic checklists is a prerequisite for this work.

SGQAB had a discussion concerning the test version of the ICES Biological Data Reporting Formats. The topic was discussed also at a joint session with the ICES/OSPAR SGQAE. Comments were given concerning the structure of the data reporting formats and possible changes were discussed and agreed within both groups. SGQAB expressed its deep concern over the delay in the delivery of the final version of the data reporting formats and data entry program and asked ICES to speed up the process of finalizing the biological data entry program. No final agreements were made, as no test versions of the data entry program were available.

SGQAB took note that, at the moment, the HELCOM database contains no biological data because ICES has not yet managed to finalize the database for HELCOM. Therefore, current periodic assessment work is being carried out based on data from local laboratories only, without general QC procedures.

After including the biological data into the ICES database, the priority task should be a proper quality check including the old data. This should be done with the option of flagging data according to the level of precision at which they can be used for assessments.

The importance of updated species lists for the HELCOM area was again stressed and the need for establishing such officially approved lists was underlined during the discussion. As the biological data entry programs fully depend on the existence of these lists, immediate actions have to be taken to provide the ICES Data Centre with these lists.

As the main activities of SGQAB can be performed only through close cooperation with the relevant groups of experts involved in several ICES Working Groups and HELCOM projects, SGQAB decided to investigate the possibilities of closer contact with relevant ICES and HELCOM bodies. The Chair will establish contacts intersessionally with Chairs of the following groups and ensure the possibility of their participation at the next SGQAB meeting. It was decided to formulate direct requests to different groups of experts to ensure the consideration of the relevant topics during their future work. The Working Groups of interest are listed below:

- ICES/OSPAR Steering Group on Quality Assurance of Biological Measurements Related to Eutrophication Effects (SGQAE);
- ICES/HELCOM Steering Group on Quality Assurance of Chemical Measurements in the Baltic Sea (SGQAC);
- Working Group on Phytoplankton Ecology (WGPE);
- Working Group on Zooplankton Ecology (WGZE);
- Benthos Ecology Working Group (BEWG);
- Working Group on Statistical Aspects of Environmental Monitoring (WGSEAM).

In establishing cooperation with the above-mentioned groups, the ACME stressed the importance of coordinating methods between the Baltic Sea and the North Atlantic.

Concerning the current status of the new Checklist of Baltic Sea Phytoplankton Species compiled by G. Hällfors, a draft version is ready to be published on the Internet, on the Alg@line website (http://meri.fimr.fi) and, if needed, to be distributed on CD-ROM to the members of the HELCOM and ICES phytoplankton working groups, with a request for constructive comments and help in finding missing data and missing articles. The final version is expected to be ready for printing in spring 2001.

It was noted that the Tenth Meeting of the HELCOM Environment Committee accepted the Phytoplankton Intercalibration for the HELCOM COMBINE Work Programme, initiated by the Alg@line Project. A questionnaire, which was sent to potentially interested laboratories, resulted in positive responses from Estonia, Finland, Germany, Latvia, and Poland. The intercalibration procedures, with sample collection and determination, are scheduled to start in March 2000 and the compilation of the results is planned for November 2000. At that time, the final report will also be made available to the Project on Quality Assurance of Phytoplankton Monitoring and SGQAB for their evaluation and synthesis.

Some laboratories working in the HELCOM area participated in the BEQUALM phytoplankton intercalibration exercise (see Section 7.3, below). The results of this intercalibration exercise will be evaluated by SGQAB.

Need for further research or additional data

The ACME emphasized that there is a continuous need for the preparation of biological reference materials, the updating of taxonomic checklists, and the organization of ring tests or intercomparison exercises.

The ACME agreed that WGPE should continue its activities concerning the collection of the results of intercomparison exercises on the use of different solvents for extraction of chlorophyll *a*.

Recommendations

ICES recommends that the present expert groups for phytoplankton and macrozoobenthos should continue their QA activities under SGQAB.

ICES recommends that a Phytobenthos Expert Group should be established for QA purposes. The primary task of the Phytobenthos Expert Group should be the compilation of a taxonomic checklist. The Phytobenthos Expert Group should also deal with the evaluation of the Guidelines and the present situation of phytobenthos monitoring activities; it should also provide comments on the ICES data reporting formats and be active in creating a data input program, as well as in all QA-related actions.

A high priority task of SGQAB should be the development of a criteria system for accepting and rejecting data for the assessment process.

ICES expresses its deep concern over the low participation at the SGQAB meetings, and inadequate allocation of resources for other QA activities carried out under SGQAB guidance. ICES stresses the importance of participation in QA activities by the Member Countries and laboratories participating in the COMBINE Programme.

7.2 Quality Assurance of Biological Measurements in the OSPAR Area

Request

Item 1.1 of the 2000 Work Programme from the OSPAR Commission: to continue to operate the joint ICES/OSPAR Steering Group on Quality Assurance of

Biological Measurements related to eutrophication parameters (chlorophyll *a*, phytoplankton, macrozoobenthos and macrophytobenthos) in order to coordinate:

- the development of quality assurance procedures;
- the implementation of quality assurance activities, e.g., the conduct of workshops and intercomparison exercises;
- the preparation of appropriate taxonomic lists of species.

Source of the information presented

The 2000 reports of the ICES/OSPAR Steering Group on Quality Assurance of Biological Measurements Related to Eutrophication Effects (SGQAE), the ICES/HELCOM Steering Group on Quality Assurance of Biological Measurements in the Baltic Sea (SGQAB), the Study Group on an ICES/IOC Checklist of Phytoplankton (SGPHYT), and ACME deliberations.

Status/background information

The ACME noted that SGQAE has prepared draft general guidelines on quality assurance for biological monitoring in the OSPAR area. These guidelines, in addition to coverage of general procedures including QA in relation to survey objectives and design, also encompass every step in sample treatment from sampling to data handling. These guidelines provide adequate detail and will be useful in designing quality assurance procedures. Guidelines for auditing quality assurance procedures will also be provided.

The degree of progress in the definition of QA procedures varies depending on the parameter concerned. So far, SGQAE has defined principles for good practice for the sampling and analysis of:

- chlorophyll *a*,
- phytoplankton,
- imaging techniques.

Effort to define good practice in the sampling and analysis of macrozoobenthos and macrophytobenthos is proceeding intersessionally. These principles will be used in defining practical and detailed guidelines for the procedures associated with these parameters. For chlorophyll *a*, phytoplankton and macrozoobenthos, intercomparison activities are being conducted under the BEQUALM project (see Section 7.3, below).

Critical QA factors and priority QA actions have been defined for monitoring chlorophyll *a*, phytoplankton, macrozoobenthos, and macrophytobenthos and guidelines are expected to be completed by 2001.

SGQAE has reviewed Standard Operating Procedures (SOPs) for the sampling and analysis of macrozoobenthos with a view to identifying examples of good practice and promoting harmonization in field sampling and laboratory analytical approaches. SOPs are the route by which national/international guidelines are translated into the working practices of individual laboratories and, as such, have a critical role to play in QA. The ACME noted that there would be benefits to reviewing the consistency of SOPs among laboratories for other biological measures.

The ACME noted the various initiatives within OSPAR towards evaluating the acceptability of data submitted by individual laboratories and recognized the importance to international monitoring effort of evolving a consistent approach to the flagging of data sets that do not fully conform with QA/AQC criteria.

In addition, the ACME noted the relevance of the recently published ICES Zooplankton Methodology Manual, which addresses methodology improvements and standardization in zooplankton studies. Although the remit of SGQAE does not formally cover zooplankton, material in this Manual can be of relevance to other types of biological measurements.

In considering the preparation of appropriate taxonomic lists of species, the ACME felt that "phytoplankton checklist" is an inappropriate term which should be replaced by "Microplankton Protists Database". It was noted that the criteria for and format of the list will be presented at the ACME Consultations Meeting in September 2000.

SGQAE reviewed the draft ICES Biological Data Reporting Formats in joint session with SGQAB, the outcome of which is reported under Section 7.1, above. SGQAE emphasized the need to update the data entry system to allow input using spreadsheet formats.

Need for further research or additional data

The ACME requested SGQAE to prepare the final draft of the guidelines for QA of biological measures in 2001, and to do so in close consultation with SGQAB.

The ACME recommends that SGQAE further evaluate criteria for judging the acceptability of biological data in international monitoring programmes with a view to producing guidelines for implementation.

The ACME recommends that SGQAE take advantage of other ongoing international QA activities in order to, after proper review, swiftly establish species checklists for macrozoobenthos and macrophytobenthos. Furthermore, the ACME stressed the importance of coordinating methods and procedures between the North Atlantic and the Baltic Sea.

The ACME recommends that, in order to apply proper quality assurance, the appropriate Working Groups (phytoplankton, macrozoobenthos, and macrophytobenthos) should organize meetings to evaluate the taxonomic validity of the species included in the database, after compilation of the list.

In order to develop an overall approach to evaluating eutrophication effects, the ACME requested WGPE and WGZE to examine the practicality and benefits of including primary production and zooplankton studies in eutrophication monitoring programmes coordinated by OSPAR.

The ACME endorsed the SGQAE concern over the generally low support provided by OSPAR Contracting Parties, which could place an unreasonable burden on core participants, despite the importance of the issues dealt with for the effective conduct of harmonized monitoring programmes at the international level. Noting also the low turn-out at the SGQAB meeting, joint discussion between the two groups led to a recommendation of a merger such that the many issues of common interest could be dealt with more effectively to the satisfaction of all Contracting Parties. At the same time, the facility to respond separately to HELCOM or OSPAR initiatives, as the need arose, would be preserved through a system of joint chairmanship.

Accordingly, the ACME requested SGQAE and SGQAB to meet jointly, with a view to exploring the practicality and benefits of a merger of the two groups.

The ACME also noted the view of SGQAE that, in its case, the focus on eutrophication issues may, for political reasons, inhibit interest from certain member countries. SGQAE therefore recommended that its remit be extended to cover QA of all biological community measures used in marine monitoring in the Northeast Atlantic area, a task that would be compatible with the proposed merger with SGQAB.

Recommendations

ICES recommends that the SGQAE remit be extended to cover QA of all biological community measures used in marine monitoring in the Northeast Atlantic area.

ICES recommends that OSPAR consider the acceptability, in principle, of merging SGQAE and SGQAB.

7.3 Quality Assurance Procedures for Biological Effects Techniques, including Fish Diseases

Request

The ACME keeps under review progress in the development of quality assurance procedures for biological effects monitoring techniques.

Source of the information presented

BEQUALM Project Newsletters 1 and 2, the 2000 report of the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO), and ACME deliberations.

Status/background information

The EU-funded BEQUALM project (Biological Effects Quality Assurance in Marine Monitoring) has now been running for over one year under the leadership of CEFAS at Burnham-on-Crouch, UK, with the objective of building an infrastructure in Europe that can provide QA/QC for most of the biological effects methods used in major marine monitoring programmes. It is particularly focused on the OSPAR Joint Assessment and Monitoring Programme (JAMP), covering all but one of the JAMP biological effects techniques. Full details of progress can be found on the BEQUALM website (http://www.cefas.co.uk/bequalm), and a summary is given below.

a) Water and sediment bioassays

This work covers tests with five species: water bioassays—using oyster embryo (*Crassostrea gigas*), a copepod (*Tisbe battagliai*), and a juvenile fish (*Scopthalmus maximus*); sediment tests—using an amphipod (*Corophium volutator*) and a polychaete (*Arenicola marina*). Laboratories from over ten countries are participating in the development programme led by CEFAS at Burnham-on-Crouch. In particular, a five-day workshop has been held to allow participants to learn and compare methods and protocols. Agreement was reached on suitable QA/QC requirements, and on appropriate reference compounds. An intercalibration exercise is scheduled to be held in the first half of 2000 in order to test the protocols in the laboratories of the various participants, and to characterize the degree of interlaboratory variability.

b) Metallothionein (MT) measurement in fish

Metallothionein is a biomarker of exposure to some heavy metals. Participating laboratories from eight countries are involved with BEQUALM. The BEQUALM programme has held a two-day workshop to identify QA requirements and initiate an intercalibration programme for MT determination in cod, dab, and flounder tissues. It was agreed that in the first intercalibration round all laboratories were free to use their own analysis technique. However, the leading laboratory (NIVA) has prepared and distributed standard operating procedures (SOPs) for the following: differential pulse polarography (DPP), enzyme-linked immunosorbent assay (ELISA), and sample preparation (subcellular separation into S-50 fractions). The ELISA and DPP techniques were demonstrated and discussed at the workshop. The first intercalibration round involved twelve laboratories from eight countries, which received calibration tissues from experimentally exposed, naturally exposed, and naturally "unexposed" fish. The results of this intercalibration will soon be available, and the next round is scheduled for August 2000.

c) ALA-D (δ-aminolevulinic acid dehydratase) in fish

ALA-D inhibition in fish blood indicates exposure to lead. There is relatively little interest in this technique, but four European laboratories have so far signed on to the BEQUALM programme. A two-day training course on the analytical method has been held by NIVA in Oslo, and a standard operating procedure has been distributed. Homogeneous blood samples from experimentally exposed and unexposed cod (*Gadus morhua*) and flounder (*Platichthys flesus*) have been prepared and will be distributed to the participants after the workshop.

d) DNA adducts in fish

The formation of DNA adducts by genotoxins such as some PAHs is an indicator of potential DNA damage. Nine participating laboratories, eight from Europe and one from North America, in total representing seven countries, are involved in the BEQUALM programme on DNA adduct measurement using the ^{32}P-postlabelling technique. In 1999, five different standard samples derived from fish were prepared and checked for homogeneity. Since DNA adduct measurement involves a multi-step procedure, the different samples were prepared to answer questions concerning different parts of the methodology. The first round of analyses was completed in October 1999. A form with a number of methodological details, concerning the way to run the ^{32}P-postlabelling technique, was also filled in by the participating laboratories. The outcome was summarized and the results were discussed during a workshop in late October 1999 which was held at the Institute of Applied Environmental Research, Stockholm University, Sweden.

The time schedule for processing the rest of the samples will extend throughout 2000 until January 2001. An additional workshop will probably be arranged, preferably before the last round in this project. The objective will be to gather together analysts from a large number of participating laboratories.

e) EROD activity in fish

EROD induction is a marker for activity of the cytochrome P4501A1 enzyme system which is induced by exposure to planar PCBs and PAHs. A total of twenty participating laboratories are involved in the BEQUALM EROD programme. A workshop was held by Fisheries Research Services (FRS) in Aberdeen early in 2000 to explore practical aspects of EROD determination and quality assurance procedures, and standard induced fish liver samples have been sent out for analysis.

f) Imposex and intersex in marine snails

This is a joint activity with QUASIMEME, and involves the imposex and intersex responses to tributyltin of *Nucella lapillus* and *Littorina littorea*, respectively. The second interlaboratory exercise on imposex and intersex was held by FRS in Aberdeen in July 1999. Eleven laboratories requesting samples returned data. Of these laboratories, nine returned data for both imposex and intersex analyses. Two returned data for imposex determination only.

All participating laboratories reported satisfactory data for Vas Deferens Sequence Index (VDSI) in *Nucella*. Differences in penis length measurements between laboratories were largely random and only three laboratories reported unsatisfactory data for Relative Penis Size Index (RPSI) (based on Z-scores calculated from the variance of the reference laboratory data). One laboratory reported significantly different results from the reference laboratory for sex determination in *Littorina*, and all except one laboratory reported satisfactory data for Intersex Sequence Index (ISI). Some laboratories reported difficulties in prostate identification.

A workshop on imposex and intersex in marine snails was held at the Marine Laboratory, Aberdeen, UK in November 1999. The workshop was held for three days, covering formal presentations, discussions and practical sessions. Four species were examined: *Nucella lapillus*, *Littorina littorea*, *Buccinum undatum,* and *Neptunea antiqua*. Discussion sessions were held to review the problems and difficulties that had been encountered and matters that had arisen from the exercise. The workshop made a number of recommendations and action points, which will be followed up. It was agreed that there would be another exercise in spring 2000 and a workshop in autumn 2000.

g) Lysosomal stability in mussels

Lysosomal stability is a useful method for measuring the effects of general chemical stress in mussels and other organisms. The particular technique being used in the BEQUALM programme is the neutral red retention (NRR) assay. Its reproducibility is being tested by sending blue mussels (*Mytilus edulis*) to participating laboratories around Europe. These animals were collected from a clean site and then exposed to a suite of chemical stressors at Plymouth Marine Laboratory. They were exposed to a mixture of contaminants, to simulate better the real world. They are identified by a code and are therefore being examined "blind"; this will remove any operator bias. In addition to control and exposure groups, participants will also be sent a further sample of mussels collected from the reference site, at the end of the contaminant exposure, to provide an additional field control. A number of laboratories around Europe have been identified to take part in this exercise. Mussels were sent out to participants in early November 1999, after trials took place at Plymouth to optimize the stress-free transportation of mussels and QA the experimental design. Participants either had previous experience with the NRR technique or were given training prior to November. After the initial study, further samples will be delivered to test the reproducibility of the technique over time.

h) External diseases of fish and fish liver histopathology

This methodology focuses on the detection of grossly visible external fish diseases considered to be linked to contaminants and toxicopathic non-neoplastic and neoplastic (including pre-neoplastic) liver lesions in flatfish. Particularly for histopathological liver lesions, there is a need for international standardization and intercalibration in the collection, processing, examination, and reporting of findings from monitoring programmes to ensure consistency and comparability in interpretation of results from various countries. This requirement has been recognized and is covered under this work package of BEQUALM. The activity includes the preparation of laboratory reference materials, the conduct of a workshop to establish protocols, hold practical exercises, and agree acceptable limits of variation, and the establishment of an intercalibration programme.

A three-day workshop was held at the CEFAS Weymouth Laboratory in October 1999 to address these points. A total of twelve scientists participated, representing eight European countries. The main objectives of the workshop concerned discussions aimed at reaching agreement on the preparation of protocols for the identification of externally visible diseases and macroscopic lesions, dissection and sampling of tissues, fixation, histological processing, sectioning, staining, and criteria for disease diagnosis. In addition, a major emphasis of the workshop was to undertake training for the identification of specific liver lesions (including the distribution of reference materials) and to agree on the format and scope of the ring test planned for later in 2000.

As requested by ICES, the workshop proposed a modification of the ICES Environmental Data Reporting Formats for the incorporation of fish liver histopathology data into the ICES Environmental Data Centre (see Section 20.5, below).

i) Fish reproductive success

This involves the brackish water and estuarine fish species viviparous blenny (*Zoarces viviparus*), whose reproductive success can be easily measured by investigating pregnant females. The method is included in the HELCOM COMBINE and OSPAR JAMP monitoring programmes. The objectives of the workpackage are to develop guidelines and databases and to perform intercalibrations. The BEQUALM programme is also a platform for future collaboration among institutes engaged in coastal fish monitoring. During the first year, new guidelines were developed. A detailed manual on how to collect and handle fish samples and how to perform the analyses of pregnant females and their broods is now available. The guidelines are connected to a database in Access format, which is also available for participating institutes.

A six-day training workshop was held at the Institute of Coastal Research field laboratory at Forsmark, Sweden, on the coast of the southern Bothnian Sea, in November 1999. Scientists from five countries participated in various activities during this week. Fish had been sampled prior to the workshop from an area in the southern Bothnian Sea. The fish were used for an initial introduction into the practical laboratory exercises: killing the fish humanely, estimating female length and weight, dissection procedures, estimating brood weight, fry malformations, survival, and individual fry length. Fry samples were preserved for determining sex.

An intercalibration was performed after the introductory exercises. This intercalibration was not only a test of the accuracy of the method, but also of the way in which skilled laboratory personnel would perform when first encountering a new technique. A sample of 35 pregnant female blenny was presented to the group. After killing the female, the brood was dissected and all fry were killed. The intercalibration covered female length, fry length, and the identification of unhatched eggs, retarded fry, and malformed fry. Every participant analysed all 35 broods. The number of fry per brood varied from 9 to 50, depending upon the size of the female. There were small differences between participants when estimating female lengths, except in one case probably due to an error when entering data into the database. Estimation of fry length (in 2.5 mm length groups) produced rather consistent results. In some broods, the estimated average length did not differ between readers. The maximum difference was one length class. No reader tended to deviate systematically from the others, and the variations seemed to be random. When comparing the results with anticipated growth retardation in effluent-exposed areas, these variations should not cause significant interpretation problems.

j) Benthic community analysis

This is an important technique for estimating the impact of anthropogenic factors on benthic invertebrate communities. The techniques involve sediment sampling and sieving, followed by identification and enumeration of the fauna, and finally the preparation of descriptive community statistics. Although these techniques have been used for many years, proper quality control of the procedures is still being developed, and the BEQUALM project is making a contribution to this.

The first benthos taxonomic workshop was held at the Zoological Museum in Copenhagen in 1997, and was a spin-off of and recommendation from the activities of the

ICES/HELCOM Workshop on Quality Assurance of Benthic Parameters in the Baltic (ICES, 1994). As one of the activities of this QA workshop, a ring test was performed with pre-sorted samples. The results revealed considerable inconsistencies regarding the taxonomy of invertebrates in the western part of the Baltic Sea. Based on ICES C.Res. 1996/3:4, the recommended workshop was organized. This was the first in an intended series of taxonomic workshops. Twenty participants from five countries (Denmark, Sweden, Finland, Germany, and Poland) attended this workshop; they were from both national institutes and private enterprises. This was intended when the workshop was first conceived. The main taxonomic groups tackled at this course were polychaetes, sponges, and selected molluscs.

A second benthos taxonomic workshop, sponsored partly by HELCOM and partly by BEQUALM, was held at the Zoological Institute and Museum in Hamburg, Germany in February 2000. The Zoological Institute and Museum at the University of Hamburg is one of the leading institutions in invertebrate taxonomy in Germany. The main topic at this workshop was crustaceans (isopods, amphipods, decapods, and some small exotic groups).

A BEQUALM macrobenthos ring test will be performed in the near future. Samples from the southern North Sea, pre-sorted, identified by experts and remixed into sterile sediment, will be distributed to twenty participants, and both sorting efficiency and taxonomic identification skills will be tested.

k) Phytoplankton assemblages

The goal of the phytoplankton intercalibration exercises within the framework of BEQUALM is to elaborate a common basis of phytoplankton counting methodology all over Europe. For this purpose, a ring test was organized in which 34 laboratories from fourteen European countries participated, representing all European marine provinces: the North Sea, the Baltic Sea, the Atlantic, and the Mediterranean (see Sections 7.1 and 7.2, above). Four common phytoplanktonic algae, grown in laboratory culture, were mixed in three different proportions, simulating different trophic conditions. The three bottles were sent out to the participating laboratories in November 1999. The laboratories were asked to identify, count, size, and calculate the biomasses of the species in each sample; however, the emphasis with this first ring test was on counting accuracy. An enclosed questionnaire allowed a detailed overview of the sampling, sample handling, fixation, storage, and processing procedures used by the participants in the different European regions. A test for the homogeneity of all samples sent out to the participating laboratories is being conducted by a third party investigator specializing in phytoplankton analysis.

A workshop on the results of this first phytoplankton ring test was held in April 2000 at the Research and Technology Centre (FTZ) of Kiel University in Büsum, Germany. The workshop included the presentation and discussion of the results of the ring test, as well as the presentation and discussion of different counting methods, including biomass determinations. Fresh and fixed phytoplankton samples were analysed during the workshop to demonstrate the respective taxonomic skill levels.

l) Chlorophyll *a* determination

This exercise was conducted collaboratively with QUASIMEME. Chlorophyll *a* is the principal phytoplankton pigment, which can be measured relatively easily (see Section 7.1). This makes it suitable to serve as a proxy for phytoplankton biomass. For the intercalibration using field samples, parallel water samples from two locations in the Wadden Sea and offshore near Helgoland were filtered on a cruise in September 1999. The filters were stored at −80 °C until shipping on dry ice to the QUASIMEME office in Aberdeen to be distributed to the participants. Filters for a total of 63 laboratories were prepared. A homogeneity analysis was performed with extra filters of this material, based on a calibrated standard high performance liquid chromatography (HPLC) technique, to ensure that discrepancies above a certain level are true discrepancies between the laboratories.

A joint QUASIMEME/BEQUALM workshop on the evaluation of the results was held in May 2000 at the FTZ. The programme for this workshop was arranged in cooperation with the QUASIMEME office and encompassed theoretical considerations concerning pigment measurements, with emphasis on chlorophyll *a*. The different methods in use for chlorophyll *a* analysis were discussed and, partly, demonstrated. Intercalibration exercises included spectrophotometry, fluorometry, and the HPLC technique to analyse phytoplankton pigment samples. Other demonstrations and presentations included flow cytometry and specific fluorescence methods to account for different phytoplankton pigments.

Noting the above information, the ACME expressed its approval of the progress being made with the development of QA/QC procedures for biological effects monitoring techniques, and reiterated the importance of this work for future marine monitoring exercises.

Recommendation

ICES notes that the BEQUALM project is proceeding according to plan, and recommends that OSPAR and HELCOM consider how best to integrate BEQUALM results into their marine monitoring programmes as soon as the project has completed its work in 2001.

Reference

ICES. 1994. Report of the ICES/HELCOM Workshop on Quality Assurance of Benthic Parameters. ICES CM 1994/E:10.

7.4 Quality Assurance of Chemical Measurements in the Baltic Sea

Request

Item 1 of the 2000 requests from the Helsinki Commission: to coordinate quality assurance activities on biological and chemical measurements in the Baltic marine area and report routinely on planned and ongoing ICES intercomparison exercises, and to provide a full report on the results.

Source of the information presented

The report of the Second ICES/HELCOM Workshop on Quality Assurance of Chemical Procedures for the COMBINE and PLC-4 Programmes (WKQAC), the 2000 reports of the ICES/HELCOM Steering Group on Quality Assurance of Chemical Measurements in the Baltic Sea (SGQAC) and the Marine Chemistry Working Group (MCWG), and ACME deliberations.

Status/background information

The ACME reviewed the work of the ICES/HELCOM Steering Group on Quality Assurance of Chemical Measurements in the Baltic Sea (SGQAC) and the results of the Second ICES/HELCOM Workshop on Quality Assurance of Chemical Procedures for the COMBINE and PLC-4 Programmes, noting further progress by SGQAC in the development of Guidelines for the Cooperative Monitoring in the Baltic Marine Environment (COMBINE) Programme. This includes finalization of the following:

- Technical Notes on Units and Conversions;
- Technical Notes on the Determination of Salinity and Temperature;
- Technical Notes on the Determination of Co-factors.

In addition, revisions and updates were prepared of existing guidelines on sampling for chemical analysis (Annex B-5 of the COMBINE Manual) and external quality assessment (Section B.6 of the COMBINE Manual). The complete COMBINE Manual is available at http://www.helcom.fi/ec.html on the HELCOM website.

Further developments needed include preparation of the following documents:

- technical notes on the determination and documentation of the measurement uncertainty of analytical methods;
- guidelines for quality control and metadata;
- technical notes on the determination of PAHs in sea water;
- technical notes on the determination of organohalogen compounds in sea water and sediments;
- sampling for chemical analysis.

The ACME expressed its appreciation for the up-to-date development of these comprehensive QA guidelines. However, for the sake of efficiency and integration with other regions, the ACME advises SGQAC to review already existing OSPAR Guidelines as well as ICES recommended analytical methods to consider whether they can be adopted for the COMBINE Monitoring Programme. This is particularly relevant for the determination of trace contaminants in marine sediments and in marine biota, where specific Baltic conditions have no effect on analytical procedures. Furthermore, the ACME recommends the adoption of the ICES Guidelines for the Determination of Chlorobiphenyls in Sediments (ICES, 1996), the Analytical Methods on the Determination of Polycyclic Aromatic Hydrocarbons (PAHs) in Sediments (ICES, 1997), and the Analytical Methods on the Determination of Polycyclic Aromatic Hydrocarbons (PAHs) in Biota (ICES, 1999).

The ACME also noted that the Guidelines on Quality Assurance of Chemical Measurements, as developed by SGQAC, will be published in the *ICES Techniques in Marine Environmental Sciences* series in 2001.

The ACME noted the success of the Second ICES/HELCOM Workshop on Quality Assurance of Chemical Procedures organized for the HELCOM COMBINE Monitoring Programme and the HELCOM Fourth Pollution Load Compilation Programme (PLC-4). This activity should be repeated not only owing to still existing differences in technical and analytical levels among the various Baltic countries, but also because the ACME is convinced that this activity is particularly important for the integration of marine, coastal, and terrestrial input monitoring programmes. This activity will assist in the development of harmonized marine and terrestrial input monitoring programmes, as well as in future integrated environmental assessments.

Recommendations

The ACME recommends that the Technical Notes on Units and Conversions, the Technical Notes on the Determination of Salinity and Temperature, and the Technical Notes on the Determination of Co-factors, as well as advice on external quality assessment should be transmitted to the Helsinki Commission for inclusion in the COMBINE Manual.

The ACME supported SGQAC's proposal that certified reference materials (CRMs) should be provided to HELCOM laboratories involved in monitoring activities for a reduced fee or, in some cases, with the fee for purchase waived.

The ACME also recommends that the scope of the work of SGQAC be expanded to include other parameters that are or might be included in the HELCOM monitoring programme, e.g., organic carbon and tributyltin.

Additional comments

The ACME supports the idea that information on specific analytical problems encountered in coastal monitoring (e.g., large amounts of suspended matter in water samples, enrichment of humic materials, low salinity, very high concentrations of nutrients, occasional oxygen hypersaturation during blooms, etc.) should be included in the existing technical notes for COMBINE.

References

ICES. 1996. Report of the ICES Advisory Committee on the Marine Environment, 1996. ICES Cooperative Research Report, 217: 100–104.

ICES. 1997. Report of the ICES Advisory Committee on the Marine Environment, 1997. ICES Cooperative Research Report, 222: 118–124.

ICES. 1999. Report of the ICES Advisory Committee on the Marine Environment, 1998. ICES Cooperative Research Report, 233: 230–237.

7.5 Certified Reference Materials for Organic Compounds for Use in Marine Monitoring

Request

This is part of continuing ICES work on quality assurance related to chemical determinations of contaminants and reporting of the results and their implications for monitoring programmes of OSPAR and HELCOM.

Source of the information presented

The 2000 report of the Marine Chemistry Working Group (MCWG) and ACME deliberations.

Status/background information

At its 1998 meeting, MCWG prepared tables of information on certified reference materials (CRMs) that provided an overview of information on reference materials currently available for use in the chemical determination of a wide range of organic contaminants in routine marine environmental monitoring, covering biota tissues and sediments. The tables were primarily based on marine matrices and only included CRMs derived from the freshwater environment where alternatives were not available or the material was of particular interest. The tables included information on the type of matrix (e.g., freeze-dried mussel tissue), concentration ranges, and associated uncertainty.

These tables were reviewed and discussed by the ACME in 1998, who endorsed the views of MCWG on the relevance and usefulness of this compilation work: in recent years both the production of and demand for CRMs for analytical determinations in different matrices and various appropriate ranges of concentrations have shown a continuous increase, in parallel with an increase in the generation of lists of chemicals of interest or concern within research and monitoring programmes. It is well known that the decision to include these new substances (generally difficult and costly to analyse) in routine monitoring usually depends on having QC/QA procedures in place and it is here that the availability of CRMs plays an important role.

In addition, the assessments of national and international monitoring data highly depend on the availability of QA information which, in the best case, includes the reporting of control chart data on CRM analyses along with their data sets by the participating laboratories (e.g., for OSPAR JAMP). Therefore, the ACME included the CRM tables in its 1998 report and agreed that MCWG should continue to update these tables periodically (e.g., annually) (ICES, 1999).

The ACME reviewed the update of the CRM tables carried out by MCWG in 2000. The following comments prepared by MCWG apply to these tables:

- The compiled tables are for information. Although every effort has been made to ensure that these tables are accurate, users of CRMs should consult vendors for full and accurate information.
- Certified calibration materials and standards are not included.
- These tables do not purport to be complete and all the CRMs listed may not be commercially available.
- Methyl mercury is not considered as an organic contaminant for the purposes of this list.

In addition, the following websites providing information and details on CRMs can be visited:

NIST	http://ois.nist.gov/srmcatalog
NRC	http://www.ems.nrc.ca, http://www.imb.nrc.ca/crmp_e.html
LGC	http://www.lgc.co.uk, http://www.vam.org.uk
BCR	http://www.irmm.jrc.be/rm/catalogue.html
NWRI	http://www.cciw.ca/nwri/nwri.html
CIL	http://www.isotope.com/newcat.htm
NIES	http://www.nies.go.jp

NIST = National Institute of Standards and Technology, USA
NRC = National Research Council of Canada; Institute for National Measurement Standards
LGC = Laboratory of the Government Chemist, UK
BCR = EC Institute of Reference Materials and Measurements—Joint Research Centre, previously Bureau of Community Reference
NWRI = National Water Research Institute, Environment Canada
CIL = Cambridge Isotope Laboratories, USA
NIES = National Institute for Environmental Standards, Japan

In the discussion of the tables by MCWG, it was noted that there are some reference (non-certified) values for NIST materials not yet included in the tables, which will be included in future updates. It was also noted that NIST SRM 1945 (whale blubber) may no longer be available, and there is no other marine mammal-based CRM known to MCWG to replace it; the difficulties previously experienced by European scientists in acquiring this material due to CITES (Convention on International Trade in Endangered Species) restrictions were also highlighted.

Again, MCWG pointed out that, in spite of increased activity in the production of these materials, there is a relative lack of suitable CRMs for the marine environment, in particular, for laboratories engaged in OSPAR and HELCOM monitoring; concentration ranges for currently available CRMs are generally inappropriate, and there does not appear to be any open sea marine sediment CRMs currently available.

The availability of other potentially useful CRMs and the development of new CRMs were discussed by MCWG. In particular, the "CHRONO" project will provide two new fish CRMs, herring certified for chlorobiphenyls (CBs) and chub certified for non-*ortho* CBs. This information will also be included in future updates of the CRM tables.

MCWG agreed that the tables provide very useful information and should be updated on a regular basis. The preparation of similar tables for trace metals and nutrients was also proposed.

The ACME noted that the 2000 update is a substantial revision of the 1998 tables, and includes relevant new information on both new materials and new compounds. The ACME therefore decided that the updated tables should be annexed to its report as Annex 4.

Need for further research or additional data

The ACME supported the views of MCWG that there is a clear requirement for the development of CRMs in appropriate matrices and concentration ranges for marine environmental monitoring, both for more traditionally monitored substances, and for organic and organometallic compounds of more recent concern in marine environmental studies.

Recommendation

ICES recommends that certified reference materials (CRMs) be used in research and monitoring studies on the marine environment, as they are a very important part of the quality control (QC) process. The tables on relevant CRMs for studies of organic contaminants in marine media, compiled by MCWG and attached as Annex 4 to this report, should be regularly updated and widely distributed, making them available and accessible, e.g., on the ICES website.

Reference

ICES. 1999. Report of the ICES Advisory Committee on the Marine Environment, 1998. ICES Cooperative Research Report, 233: 47–56.

7.6 Oxygen Solubility Tables

Request

This is part of the continuing ICES work to keep under review information of interest concerning new methodologies and other items relevant for monitoring, including harmonization of data.

Source of the information presented

The 2000 report of the Marine Chemistry Working Group (MCWG) and ACME deliberations.

Status/background information

In 1986, UNESCO recommended that the equations of Benson and Krause (1984) for the calculation of oxygen solubility should thereafter be used, and earlier tables should be updated. However, thus far no new table has been published despite the fact that the values in the UNESCO International Oceanographic Tables previously published are low by 0.1 % since they are based on ideal gas molar volume instead of actual O_2 molar volume.

MCWG discussed whether it could be useful for ICES to publish such tables. With the present widespread use of personal computers, tables do not appear to be as necessary as they have been in the past. However, it is important that users can check the equations they have entered into their computer, and the only way is to offer them a limited number of solubility data covering the usual range of natural conditions. Such "tables" could, for instance, give solubility values every 1–2 °C in temperature and every 1–2 (PSS78) in salinity. A paper with the recommended equations was prepared by A. Aminot (France).

The ACME reviewed this issue and decided that the paper with the recommended equations, prepared by A. Aminot, should be annexed to its report as Annex 5. The ACME also decided that the formulas and some limited tables should be made available on the ICES website, accompanied with the necessary background material, including references, and conversion coefficients to be used. An oceanographic calculator relevant to this purpose can be found on the ICES website at http://www.ices.dk/ocean/oceancal/start.htm.

Reference

Benson, B.B., and Krause, D., Jr. 1984. The concentration and isotopic fractionation of oxygen dissolved in fresh water and seawater in equilibrium with the atmosphere. Limnology and Oceanography, 29: 620–632.

7.7 Developments within QUASIMEME and QUASH

Request

This item is an ACME initiative to follow the development in these two QA projects owing to the long-standing ICES involvement in quality assurance matters.

Source of the information presented

The 2000 report of the Marine Chemistry Working Group (MCWG), internal reports from the two projects, and ACME deliberations.

Status/background information

The ACME recalled that QUASIMEME Laboratory Performance Studies (LPS) became available to all laboratories worldwide from June 1996. QUASIMEME is open to all organizations making chemical measurements in the marine environment, for instance, in the context of national or international monitoring programmes, for individual or collaborative research, or for contract studies.

The LPS are designed to support the quality management of laboratories and the quality of measurements in general. The assessments provided by QUASIMEME may be used:

- to complement internal laboratory QA;
- to support laboratory accreditation;
- to support QA information submitted with environmental monitoring data to national or international programmes.

The third year of QUASIMEME (June 1998 to May 1999) consisted of four LPS rounds (Nos. 14–17) as a continuation from the initial EU project, which concluded at Round 5, and the previous two years of the subscription scheme. The assessments for all four rounds have been completed by an internal assessment team.

No major changes have been seen in the performance of laboratories participating in the QUASIMEME LPS. The LPS reviews are updated each year, which is sufficient. MCWG advised in detail on what figures and tables would be relevant for further LPS reviews.

The ACME also took note that during the past year there had been considerable interaction between QUASIMEME and the BEQUALM project concerning interlaboratory studies on the determination of chlorophyll *a*, as well as intersex and imposex studies (see Section 7.3).

The EU-funded Quality Assurance of Sample Handling (QUASH) project was developed as a direct response to the requirements of the OSPAR Commission, the Helsinki Commission (HELCOM), and the Mediterranean Pollution and Research Programme (MEDPOL) to establish a holistic quality management and training programme to improve the sample handling techniques used and the measurement of co-factors in marine monitoring programmes and to provide data of known quality. These international and relevant national programmes provide data on mandatory determinands for the Quality Status Reports of the Northeast Atlantic, the Baltic Sea, and the Mediterranean marine environments.

The analytical QA of data for these programmes is covered by the QUASIMEME Laboratory Performance Studies. The additional resources provided by the QUASH project focus on the improvement and control of the total uncertainty of two interrelated factors: sample handling and the measurement of co-factors. The environmental data generated by laboratories are not necessarily of poor quality, but are often of unknown quality primarily because the variances associated with sample handling and the measurement of the co-factors are not generally estimated. These two areas are considered to be the main sources of unquantified error and are currently the weakest link in the quality chain. The magnitude of the total variability is a function of the variability associated with each step from sampling to data interpretation. Since co-factors are frequently used to normalize the data on contaminants in sediment and biota, the variance resulting from the sampling and measurement of these parameters can have a significant impact on the interpretation of the data.

Poor or misused sampling methods and incorrect sample handling cause gross errors which cannot be corrected by a valid chemical measurement in the laboratory. The main co-factors currently used to normalize the mandatory determinands and to provide an assessment of the environmental impact of these contaminants are lipids for biota, and total organic carbon, iron, and aluminium for sediments. Methods which have been fully validated among laboratories are required to provide a similar level of improvement and uncertainty for the co-factors as for the determinands themselves.

Guidelines for sampling and sample handling exist, but they can differ between monitoring programmes for the same matrix-determinand combination; they have never been fully tested or evaluated for different users in broad-based international interlaboratory studies. Although the technology and methodology are available, the agreement between marine laboratories is currently lacking, due to underdeveloped quality systems, with regard to validated methods for sample handling and for the measurement of the co-factors. This situation is compounded by a lack of test materials and QC check samples to validate the sampling methods and operate a holistic QA scheme.

The QUASH project consists of the following parts:

1) Sampling and preservation of nutrients in sea water;
2) Lipids and water as co-factors in biota;
3) Sample handling of biota;
4) Sample handling and normalization procedures for sediments;
5) Preparation of test materials;
6) Laboratory and field performance studies.

QUASH has been developed as a coordinated project to allow participating laboratories to undertake a stepwise improvement programme to identify and, where possible, correct these problems. The project provides an audit trail to give (i) feedback via critical evaluation as a learning tool, (ii) a means to obtain detailed sample handling information, and (iii) control through management training and the use of key laboratory-to-field-to-laboratory test samples.

The main studies focus on the estimation and control of the uncertainty associated with sample manipulation prior to laboratory analysis. The validation of sampling methods, sample handling techniques used, and the measurement of the co-factors are central to obtaining reliable marine chemical measurements. Procedures for quality assessment and quality control of sample handling are obtained concurrently to estimate the levels of uncertainty and thus the improved reliability of data on co-factors from sampling to analysis.

The information from these studies supports the development of each laboratory's scope for accreditation for field measurements and the establishment of guidelines for the audit of field manipulations. It provides a holistic quantitative assessment to guarantee the quality of information to achieve the objectives of the marine monitoring programmes and provide recommendations to OSPAR, HELCOM, and MEDPOL with respect to guidance on sampling, sample handling, and the measurement of co-factors.

The ACME took note of the successful termination of the QUASH project which has generated much useful information as well as recommendations to regulatory commissions on procedures for sampling and sample handling.

8 MARINE CONTAMINANTS: DISTRIBUTION, TRANSPORT, AND EFFECTS

8.1 Information on Specific Contaminants

Request

This is part of the continuing ICES work to keep under review contaminants of interest in a marine environmental context.

Source of the information presented

The 2000 report of the Marine Chemistry Working Group (MCWG) and ACME deliberations.

Status/background information

New information on *tris*(4-chlorophenyl)methanol (TCPM) and *tris*(4-chlorophenyl)methane (TCPMe)

Since 1996, the MCWG has kept the development of these substances in the marine environment under review. In that year, the ACME accepted a review prepared under MCWG concerning these compounds for publication in its report (ICES, 1996).

Some new results regarding *tris*(4-chlorophenyl) methanol (TCPM) and *tris*(4-chlorophenyl)methane (TCPMe) in marine mammals were presented at the 2000 MCWG meeting. The sources of these compounds are still not known, but one theory suggests that TCPM is a contaminant in technical DDT products, and that TCPM is converted to TCPMe in the environment.

TCPM and TCPMe were measured in blubber samples of seals and whales from the Estuary and Gulf of St. Lawrence, Canada, using ion-trap mass spectrometry (MS/MS) detection. Detectable concentrations of both TCPM and TCPMe were observed in all of the samples analysed. Concentrations of these compounds varied with species, ranging from 1.7 to 153 ng g^{-1} lipid weight for TCPM and from 1.3 to 50.6 ng g^{-1} lipid weight for TCPMe. TCPM was present at from 1.3 to 10 times higher concentrations than TCPMe. The highest levels of both TCPM and TCPMe were observed in adult male beluga whales (*Delphinapterus leucas*) from the St. Lawrence Estuary, while adult female beluga whales from the same area showed levels similar to those in the seals examined. Among the four seal species investigated, TCPM and TCPMe levels were the highest in grey seals (*Halichoerus grypus*) and hooded seals (*Cystophora cristata*), and lowest in harp seals (*Phoca groenlandica*). Intermediate levels were found in harbour seals (*Phoca vitulina*). Based on limited data, concentrations of TCPM and TCPMe in the blubber of marine mammals from North America seem to be similar to those observed in samples from Russia and Asia, but about ten times lower than those seen in samples from Europe.

Ratios of both 4,4′-DDE/ΣDDT and TCPM/ΣTCP (ΣTCP = TCPM + TCPMe) were very similar between animals from the same species. Strong correlations between ΣTCP and ΣDDT were also observed for each species of mammals, most likely indicating that both ΣTCP and ΣDDT are bioaccumulated in marine mammals. It was mentioned that similar correlations were also seen with ΣCB concentrations.

Some data for TCPM and TCPMe in fish samples from Canada and Europe are also available. Both compounds were found in almost all Canadian fish samples, and the levels of TCPM were again generally higher than those of TCPMe. This has not been the case in studies in the Baltic Sea and along the Dutch and Belgian coasts, where higher concentrations of TCPMe compared to TCPM have been found. Differences in the ratio between the compounds may depend on the proximity of various sources, the trophic level and/or interferences in analytical methods.

A comparison of ΣTCP and ΣDDT in the fish samples showed no correlation between those parameters, which does not support the theory that DDT formulations are the major source for the TCPM/TCPMe compounds.

Need for further research or additional data

There is a need to obtain more information about the occurrence of these compounds in the marine environment and also to obtain information about the comparability of the analytical methods, e.g., through an interlaboratory study on the determination of TCPM and TCPMe. There is also a need for more information on the toxicity of TCPM and TCPMe as only few data are available (see also ICES, 1996).

Recommendations

ICES encourages its Member Countries to gather more information about TCPM and TCPMe in the marine environment and also to take part in interlaboratory studies comparing analyses of these compounds.

New information on volatile organic contaminants in biota

The MCWG has for a long time kept under review the development of volatile organic contaminants in the marine environment. In 1995, the ACME published reviews concerning these compounds (ICES, 1995).

Two papers presented at the 2000 MCWG meeting on studies of volatile organic contaminants in biota provided the background for the information below.

Concentration levels of twelve priority volatile organic compounds (VOCs) were reported in two species of vertebrates and four species of invertebrates from sampling stations in the southern North Sea. The analyses were performed using a purge and trap system coupled to gas chromatography-mass spectrometry (GC-MS). In general, the concentration levels of VOCs found were of the same order of magnitude as those previously reported in the literature. The concentrations of the chlorinated hydrocarbons (CHCs), with the exception of chloroform, tended to be lower than those of the monocyclic aromatic hydrocarbons (MAHs). The experimental data were statistically evaluated using both cluster and principal component analysis (PCA). From the results of cluster analysis and PCA, no specific groups could be distinguished on the basis of geographical, temporal or biological parameters. However, based on the cluster analysis and the PCA, the VOCs could be divided into three groups: C_2-substituted benzenes, CHCs, and benzene plus toluene. This division could be related to different types of sources. Finally, it was shown that organisms can be used to monitor the presence of VOCs in the marine environment. The observed concentration levels were compared with proposed safety levels; the observed average concentrations were generally much lower than the proposed safety levels.

Concentration levels of thirteen of the sixteen volatile organic compounds investigated in water and fish from Lake Mälaren in Sweden were reported. The analyses were performed using solid-phase micro-extraction (SPME). That technique was considered to be useful for the determination of these solvents in both water and biological samples. However, internal standards have to be extremely pure and background contamination is difficult to avoid in the laboratory, and may emanate from ambient air. For each compound, the fish-water partition coefficient (K_{fw}) was determined. Concentrations of volatile organic compounds determined in water were found to compare well with those inferred from the concentrations in fish. Xylenes were present at the highest concentrations, followed by chloroform and toluene. It was concluded that the overall levels of solvents in both water and biota from Lake Mälaren were low.

Need for further research or additional data

There is a need for assessing the relevance of these compounds in relation to the marine environment.

Polychlorinated dibenzo-*p*-dioxins (PCDDs)

The issue of dioxins in fish is of great concern, as highlighted by the recent Belgian dioxin crisis. In this incident, contamination was caused by used oil containing polychlorinated biphenyls. The oil also contained a high level of polychlorinated dibenzofurans (PCDFs).

Data from the Netherlands from 1992 suggest that the major contribution to the toxic equivalent (TEQ) values observed in fish was due to non-*ortho* and mono-*ortho* CBs. In general, this was a factor of 2 to 4 times higher than the PCDD/PCDF component of the TEQ values.

WHO Acceptable Daily Intake (ADI) guidelines for dioxin intake are 1–4 pg kg^{-1} body weight day^{-1}. Based on this value, there may be serious implications for future consumption of fishery products. Also, legislation introduced in Belgium subsequent to the dioxin crisis specified a limit for the sum of 7 CBs of 200 ng g^{-1} lipid for animal products with less than 2 % lipid content. Although this did not include fish, if a similar limit were to be introduced in Europe which is also valid for fish, then fishery products from even relatively pristine areas may exceed it.

In a number of European countries, few data are available on the concentrations of dioxins in fish, presumably due at least in part to the difficulty and expense of these analyses.

Need for further research or additional data

There is a need to obtain more information about the implications of consumption of fishery products in relation to contamination of the marine environment, not only for dioxin but also for other compounds, e.g., tributyltin (risks to top predators).

References

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8.2 Abiotic and Biotic Transport and Effects of Dioxins

Request

This is part of the continuing work of ICES to keep under review marine contaminants and their effects.

Source of the information presented

The 2000 report of the Working Group on Marine Sediments in Relation to Pollution (WGMS) and ACME deliberations.

Status/background information

The ACME was informed about a new Norwegian project on abiotic and biotic transport and effects of dioxins in the Frierfjord, southern Norway. This fjord has strongly been affected by dioxin discharges over several decades. A strong decline in the quantities discharged since 1991 has not been equally reflected by declines in dioxin concentrations in biota. This caused concern and the present project is intended to determine the reason for these discrepancies.

The ACME welcomed this project with special interest and hopes to receive further information on the progress in this work. A summary of the problem and the objectives and working procedures of the project are presented below.

The Frierfjord, a silled fjord in southern Norway, and adjacent coastal areas have received considerable inputs of dioxins since the 1950s from a local magnesium plant located at the inner reaches of the fjord. In the period from 1965–1975, a discharge of 5–10 kg per year has been estimated, decreasing to 1–8 g per year after 1991 calculated as tetrachlorodibenzo-*p*-dioxin (TCDD) toxic equivalents (TEQ). The discharges from the magnesium plant have consequently been a major point source even on a European scale.

The strong abatement initiatives taken during the past decade have been a consequence of the local environmental targets aiming that seafood from the area might be consumed without any restrictions by the year 2000. Although a strong decline in dioxin concentrations in the monitoring matrices has been observed, the concentrations in organisms seemed to stabilize at a lower, but still too high, level. Various hypotheses have been proposed to explain the discrepancy between discharge reductions and the concentration responses of the monitoring organisms, including accumulation from contaminated prey organisms, release from the sediments and additional, unknown sources of dioxins.

To understand the abiotic and biotic dynamics and distribution of dioxins in the fjord, a major four-year programme was initiated in March 2000. The programme will include mass balances/models for the water-sediment system, transport and effects in food chains, and ecological risk modelling.

For the water-sediment system, three different types of model calculations have been carried out so far. The first is a simple estimate of the amount presently exported from the fjord, based on the assumption that the dioxins in the sediments equilibrate with the bottom water. The second is an unbalanced steady-state box model. It uses measured concentrations in the fjord sediments and biota together with measured concentrations of dioxins in air and water. The model gives intermedia fluxes between sediment/water, water/air, fjord water/adjacent water, and also estimates the amounts in sediments, water, air, and aquatic biota. Thirdly, a time-dynamic (level 4) fugacity-based multimedia model with emissions to air and water, and advections from surrounding air and water as input sources of dioxins, will be used to simulate the build up of concentrations and intermedia fluxes in the whole of the Frierfjord drainage basin. New sampling of various matrices will be performed to improve the validation of the models, and the models *per se* will be further elaborated.

8.3 Relationship between Soft-bottom Macrofauna and PAHs

Request

This is part of ongoing ICES work to keep under review marine contaminants and their effects.

Source of the information presented

The 2000 report of the Working Group on Marine Sediments in Relation to Pollution (WGMS) and ACME deliberations.

Status/background information

The ACME took note of a Norwegian study on the relationship between soft-bottom macrofauna and PAHs from smelter discharges in Norwegian fjords and coastal waters.

The ACME highlighted that the approach in the study, using an entire species assemblage and PAHs together with other variables as explanatory and environmental variables in combination with appropriate statistical tools, is more sensitive than previously used approaches based on diversity indices to demonstrate effects of PAHs on soft-bottom macrofauna. The ACME concluded that it would be interesting to apply this approach in similar studies with other contaminants.

A summary of the objectives and outcome of the study is given below.

Background

The study of natural species communities is a key element in many monitoring programmes addressing the spatial and temporal effects of contaminants. However, the effects may be small compared with the influence of natural abiotic factors and may also be overshadowed by large biological variation. The inherent difficulty may lie in correctly linking biotic responses and environmental influences, in particular when it comes to detecting and interpreting subtle changes. Despite the complexity of the systems, many experts stress the importance of community or ecosystem studies for risk assessments because single-species studies such as bioassays may fall short in predicting the responses of natural systems.

In a programme involving sediment PAH contamination and benthic community structure in Norwegian fjords and coastal waters, the aim was to explore whether effects could be detected using elaborate statistical tools such as canonical correspondence analysis (CCA).

Previous studies have shown that effects of PAHs on soft-bottom fauna have been small. However, these studies have to a great extent been based on diversity indices. Recent research has shown that this is not a sufficiently sensitive method.

Analysis

Possible relationships were explored using the entire species assemblage and PAHs together with other variables as explanatory or environmental variables. Factors such as input of fresh water, terrestrial organic carbon, depth, etc., are known to influence species composition. These are the natural factors that determine soft-bottom community structure in fjords in general. Therefore, samples from five fjords unaffected by PAHs were included in the analysis to rule out relationships that were determined by gradients in natural factors. In fact, an equivalent number of affected and unaffected samples were included to balance the statistical analysis.

Three types of analyses were performed:

1) First, general faunal gradients and responses to environmental variables in fjords were illustrated using data from all stations.
2) Second, faunal gradients in two fjords were assessed. In distinction to the other fjords, more sediment characteristics had been quantitatively determined in these two fjords. In the first and second analyses, all environmental variables were entered in the analysis.
3) The third analysis focused particularly on species distribution related to PAHs. The statistical analyses were carried out with PAH as the only environmental variable, implying that the first CCA axis turns into a linear gradient for PAH in which species are arranged in order. And, very importantly, the explanatory variables for depth and distance from the river outlet points were entered as covariables in order to eliminate variation related to these factors. One of the main reasons for using CCA was to make use of the facilities of this technique to separate among effects of various environmental factors.

The first and second CCA should, and did, reveal the importance of the natural environmental variables both for the species assemblage and the trophic community structure. This was captured on the first and second CCA axis. However, on a higher axis, i.e., axis 3, clearer patterns emerged in relation to PAH level in the sediments. This was then further elaborated in the third CCA showing significant species and community structure trends along a PAH gradient.

Results and Conclusions

The examination of data from Norwegian smelter-affected fjords revealed that significant faunal changes pertaining to trophic composition, and to some extent to systematic groups, were identified along sedimentary PAH gradients. At high PAH levels, detritus feeders were generally reduced while carnivores seemed to be unaffected or possibly even stimulated. The changes appeared to commence at rather low PAH levels, before the total number of species and the community diversity were affected, and also before detrimental effects became obvious. Clearly, the general effects of PAHs on the communities were limited, but the functional changes may represent a type of early response signal preceding more fundamental changes evident by species reductions or reduced diversity.

In the present case, the species patterns extracted on the PAH gradient represented no more than 5–10 % of the total variance. These patterns were not observed in the traditional treatment of the species data. Hence, this type of statistical approach is a powerful tool in investigations of relationships between contaminants and effects on benthic organisms.

8.4 Assessment of the Toxicity of Dredged Marine Materials

Request

The disposal of dredged materials in the marine environment continues to be a very substantial activity, and both ICES Member Countries and international marine monitoring programmes are in need of improved procedures for assessing the risks posed by this practice.

Source of the information presented

The 2000 report of the Working Group on Biological Effects of Contaminants (WGBEC) and ACME deliberations.

Status/background information

In most countries around the Northeast Atlantic, the maintenance of harbours and ports involves dredging of very large volumes of sediments and their subsequent disposal in coastal waters (more than 100 million tonnes per year in the OSPAR area alone). This can be of environmental concern if the dredged materials are highly contaminated. Countries in Western Europe generally comply with the international guidelines of OSPAR and the London Convention regarding offshore disposal of dredged material. At present, the assessment of the dredged materials is based on chemical analyses of a standard suite of chemicals and trace metals. Following the international guidelines, lower and upper action levels are set by national authorities, which form the basis of the licensing process for offshore disposal. However, there is a general belief that chemical analyses do not provide a sufficient assessment of the environmental risks of dredgings disposal and that sediment bioassays are required as an additional tool to measure the toxicity of sediments (EPA-USACE, 1998).

The present status of the use of bioassays in dredged material assessment in the Netherlands, the United Kingdom, and France was discussed by WGBEC. Several points of agreement were identified:

a) For assessment of the toxicity of dredged materials, a total of twelve bioassays have been selected by IFREMER (France), CEFAS (UK), and RIKZ (Netherlands). The whole sediment bioassay with the amphipod *Corophium volutator* is common to all three countries and the Microtox-SP to two countries. The other bioassays (including those using the heart urchin *Echinocardium cordatum*, the lugworm *Arenicola marina*, the microalga *Phaeodactylum tricornutum*, the oyster embryo *Crassostrea gigas*, the fish *Dicentrarchus labrax*, the amphipod *Bathyporeia sarsi*, the crustacean *Artemia salina*, and the copepods *Tigriopus brevicornis, Acartia tonsa* and *Tisbe battagliai*), not all of which ultimately proved suitable for dredged material assessment, were selected on the basis of preferences by the national laboratories and the availability of standard operating procedures. It was agreed that considerable effort should be placed on quality control and quality assurance of these tests, including interlaboratory comparisons such as those taking place in the BEQUALM programme (Murk *et al*., 1996; Stronkhorst, 1998; Quiniou and Alzieu, 1999; Roddie and Thain, 2000; Thain and Bifield, 2000).

b) The dioxin-responsive chemical-activated luciferase gene assay (DR-CALUX), and similar genetically engineered cell lines, have a special application as they identify dioxin-like compounds that are a major group of concern for the marine environment. The DR-CALUX can be applied to extracts of sediments, pore waters, biota, and other matrices.

c) There appear to be some variations in the exact methodology used. For instance, in the case of the oyster embryo bioassay, there are differences in methods of exposure which lead to differing results in various laboratories.

d) *Tisbe* and *Leptocheirus* are presently used in acute tests, but chronic endpoints are also available.

e) The pros and cons of the Microtox-SP (solid phase) test have been discussed. The silt and clay content of the sediment appears to be a major confounding factor in this assay which could be corrected for when the grain size fraction is considered. There is some disagreement as to the validity and merit of this approach and the issue has not been resolved.

f) The contamination of harbour sediments with TBT is a major issue in most countries. Recently, a specific bioassay which may be applicable in this context has become available that identifies the development of imposex in the snail *Hinia reticulata*.

g) The merits of determining lethality or other effects in a whole sediment test by measuring an L(E)C50 (after producing a dilution series of dredged material with reference sediments of different physical and chemical characteristics—organic carbon content, redox, etc.) needs to be resolved, but this approach might be useful in the future because it gives toxicity values with confidence limits.

h) The use of a test-battery is perceived to represent best practice since different species can respond in different ways to the same dredged material.

i) Theoretical assessment criteria are used in all three countries and differ substantially in their underlying approach. In the case of *C. volutator*, for instance, 40 % mortality after ten days has been applied by CEFAS as an arbitrary criterion; 25 % mortality after ten days has been set as an ecologically relevant level and made operational on the basis of a power analysis by RIKZ, while IFREMER uses the ten-day LC50 as its operational criterion.

j) The use of sediment elutriates (to mimic the exposure in a dredged material plume during dispersion) and pore water is a possibility. The relevance of pore water tests in comparison to the interpretation of whole sediment exposure is debatable. No consensus has been reached on this point.

k) Bioaccumulation tests are not at present applied in the UK, France or the Netherlands. However, the transfer of persistent compounds through the food chain is a relevant topic to consider when licensing for dredged material disposal. There are possibilities for modelling this phenomenon, but bioavailability remains a difficult issue and might require the development of test systems with, for instance, deposit feeding bivalves like *Scrobicularia plana*, *Macoma balthica*, *Abra alba*, and *Nucula* spp.

l) It is apparent that biological testing will become part of the assessment framework for contaminated dredged material. What needs to be determined now are the trigger points for the inclusion of biological effects measurements. Preferably, international guidelines should be developed that clarify the triggers for biological testing.

m) Of the twelve techniques that were presented, there was only one bioassay that incorporated a biomarker to indicate exposure (*inter alia,* to organophosphorus and carbamate pesticides, i.e., the inhibition of cholinesterase in the copepod *Tigriopus brevicornis*). There was considerable interest in the wider use of such an approach as it potentially increases test sensitivity and can be specific for the group of compounds of interest.

The ACME agreed on the following points:

1) Validated bioassays are required in cases where toxicity assessment of dredged material is relevant and supplements chemical analysis.

2) The selection of appropriate bioassays is driven by costs, ecological relevance, time constraints, sensitivity, robustness, and local preferences.

3) Analytical chemistry alone can be used to define an upper cut-off point above which sea disposal should not occur.

4) Assessment criteria should generally be defined in terms of biological significance, and not solely by statistical significance.

5) Because of the differences in ecotoxicological responses to chemicals among species, the use of a suite of bioassays is needed.

6) In addition to the use of bioassays, there is a need to assess the overall impacts at disposal sites, e.g., by investigations of benthic community structure.

7) Great care is needed with sediment sampling, including standardization, to obtain representative samples.

8) There is a need to account for the presence and possibly the effects of persistent and/or bioaccumulative substances.

9) There is a need to develop the use of biomarkers in dredged material bioassays in order to increase sensitivity and contaminant-specificity.

Need for further research or additional data

The following research questions need to be addressed:

1) Is it possible to define a lower chemical cut-off point that conservatively triggers the application of dredged material bioassays?

2) Do chronic bioassays add useful knowledge in dredged material assessment?

3) In which cases are bioaccumulation tests with dredged material required, or is partition modelling sufficient?

4) Do bioassays with elutriates and pore waters add useful information to that obtained from whole sediment tests?

5) Is there value in Toxicity Identification and Evaluation (TIE) techniques for evaluating causes of toxicity (Burgess *et al*., 1996)?

References

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8.5 Workshop to Evaluate the Utility of Artificial Intelligence Procedures in the Assessment of Pollution Effects in Flatfish

Request

This is part of continuing ICES work to improve the tools available for monitoring and interpreting the biological effects of contaminants in the marine environment.

Source of the information presented

The 2000 report of the Working Group on Biological Effects of Contaminants (WGBEC) and ACME deliberations.

Status/background information

The ACME was informed that WGBEC has reviewed progress with the organization of an ICES workshop, tentatively planned for spring 2001 in Seattle, WA, aimed at the investigation of artificial intelligence (AI) approaches for integrating complex biological and chemical monitoring data into more holistic marine environmental assessments. The workshop will attempt an evaluation of candidate techniques, possibly using the extensive chemical (PAHs), fish biomarker, and liver disease data gathered by the U.S. National Oceanic and Atmospheric Administration (NOAA) from Puget Sound on the US west coast. Contacts with NOAA have been made in order to explore the possibility of running an evaluation workshop in Seattle that could be held back-to-back with the 2001 meeting of WGBEC.

Unfortunately, NOAA does not yet have the entire Puget Sound database gathered together in a convenient form. Furthermore, it appears that resource limitations may prevent this from being done in the near future, so a meeting in Seattle in spring 2001 will be difficult or impossible to arrange.

However, there is a continuing need for, and a desire to proceed with, evaluation of integrative AI methods. Furthermore, there may now be several other suitably comprehensive data sets which an AI workshop might be able to use. These are listed below:

- data on contaminants and their effects on fish at other sites in the USA;
- data on fish in the Canadian National Contaminants Information System;
- data on a range of fish biomarkers and contaminants being generated in an ongoing programme by NIVA in Norway;
- data on dab held by CEFAS, covering eighteen stations in UK waters sampled in each of two successive years. Variables include EROD, PAH metabolites in bile, DNA adducts, liver histopathology, external disease, and metals and PAHs in sediment.

The ACME noted that WGBEC will investigate the suitability and availability of these data sets during the coming year.

The ACME agreed that efforts should continue with the organization of a workshop for the evaluation of artificial intelligence techniques applied to marine monitoring data.

9 FISH DISEASE ISSUES

9.1 New Trends in Diseases of Wild and Farmed Fish and Shellfish Stocks

Request

This is part of continuing ICES work to consider new developments with regard to fish and shellfish diseases.

Source of the information presented

The 2000 report of the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) and ACME deliberations.

Status/background information

The ACME reviewed the relevant sections of the WGPDMO report providing information on new trends in the occurrence of diseases in wild fish stocks based on national reports from ICES Member Countries.

Special attention was drawn to new trends in the distribution of the following diseases in wild and farmed fish and shellfish stocks:

Viral Erythrocytic Necrosis (VEN) is widely distributed in herring *(Clupea harengus pallasi)* in Alaska at moderate intensities. However, it is causing significant mortalities in young-of-the-year herring in this area.

Anisakis simplex ("herring worm") in herring in the Baltic Sea has mainly affected the spring-spawning stocks in the southwestern Baltic. In recent years, it has shown an increasing trend in the spring-spawning herring population from the spawning areas of the southeastern Baltic Sea, with prevalences of up to 63 % in the Gdansk Basin.

The prevalence of **liver nodules** in North Sea dab (*Limanda limanda*) has shown a decreasing trend for the past 4–5 years. As this lesion has been associated with chemical contamination, it may be an indicator of a general improvement. However, more research is needed before any conclusions can be drawn.

Epitheliocystis-associated gill damage caused by rickettsia- or clamydia-like organisms is an increasing problem in salmon farming in some parts of Norway. The disease has been reported to affect salmon three to six months after their transfer to the sea. Mortalities can reach 10–30 % over a period of two to three months.

Paramoeba**-**associated mortalities of lobsters (*Homarus americanus*) were reported from commercial traps in western Long Island Sound, New York, and Connecticut waters. The mortalities began during September 1999, with approximately 8 % of lobsters found dead in the traps, and mortalities rising to 30 % after two days. Nervous system infection by an organism identified as a *Paramoeba* sp. is thought to be the causative agent. At the end of the lobster season, all lobsters in the area appeared to have died, as no catches were made in the traps. Laboratory investigations showed that all lobsters were infected by the *Paramoeba* sp. However, it is still not clarified as to whether the *Paramoeba* sp. is the causative agent or it appears in the lobster due to an unknown stress situation.

Haemic neoplasia in soft-shell clams (*Mya arenaria*) was associated with a serious mortality and complete loss of one cultured bed in Prince Edward Island. This is the first record of prevalences up to 95 % and mortalities associated with this disease in Atlantic Canada.

Need for further research or additional data

The development of the new disease trends mentioned above should be followed by ICES Member Countries in the coming years. Especially the *Paramoeba*-associated mortalities of lobsters should be considered, as this disease seems to have a considerable impact on the fishery along a relatively long coast line.

9.2 Progress in Work on Fish Disease Data Assessment

Request

This is part of continuing ICES work to develop and apply appropriate statistical methods to analyse and assess fish disease data held in the ICES Environmental Data Centre in combination with other types of relevant data in ICES data banks.

Source of the information presented

The 2000 report of the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) and ACME deliberations.

Status/background information

During its 1998 and 1999 meetings, the ACME reviewed the results of WGPDMO work to analyse data on the prevalence of fish diseases held in the ICES Environmental Data Centre. Data on the occurrence of externally visible diseases (lymphocystis, epidermal hyperplasia/papilloma, and acute/healing skin ulcerations) in common dab (*Limanda limanda*) and European flounder (*Platichthys flesus*) from the North Sea and adjacent areas, including the Baltic Sea, were extracted and tested for the presence of significant spatial and temporal long-term trends (ICES, 1999). In a subsequent pilot study, parts of the disease data were analysed in combination with other types of ICES data (contaminants, nutrients, oceanography, and catch per unit effort (CPUE)), in order to identify relationships between the disease prevalence and potential explanatory factors (ICES, 2000) and to assess the potential to carry out further analyses.

In the discussion of the results of the pilot study, the ACME emphasized that the results were promising since a number of significant relationships between the disease prevalence and environmental factors included in the analysis could be identified. It was considered worthwhile to investigate these relationships in more detail using an extended data set covering larger geographical areas and time spans. However, it was also noted that there was an apparent lack of non-fish-disease data (in particular with respect to contaminants in sediments and biota) in the ICES data banks, making a more comprehensive analysis difficult. The ACME, therefore, recommended at its 1999 meeting that ICES Member Countries should be encouraged to enhance their efforts to submit historic and current data held in national data banks to the ICES data banks in order to facilitate a more comprehensive data analysis.

At its 2000 meeting, WGPDMO reviewed a report produced intersessionally providing results of the continued analysis, using an extended set of data and applying a modified statistical procedure taking into account the problems encountered by using interpolated values (see Section 6.6.2). The data extension was largely due to the fact that the data availability policy of the ICES Oceanographic Data Centre has been modified recently and that historic data until 1989 are now directly available via the Data Centre as public domain. With regard to the submission of new contaminant or biological effects data to the ICES Environmental Data Centre, WGPDMO noted with concern that no progress has been made in the past year. A new statistical approach was developed for a more appropriate way to incorporate interpolated values in the analysis. This method allows the statistical consideration of extra variances caused by the use of interpolated instead of measured values.

The results of the statistical analysis based on the above modification of the use of interpolated values showed that, in the univariate analysis, the factors identified as having a significant relationship with the disease prevalence (e.g., water temperature and salinity, certain contaminants in sea water, sediments or tissues) were identical to those identified in the previous analysis (ICES, 2000), with only a few exceptions. However, significance levels were generally considerably weaker. The results of the multivariate analysis involving all potentially explanatory factors (oceanographic, nutrient, and contaminant data as well as CPUE data) indicated that, even after applying stricter rules for the incorporation of interpolated values, significant relationships continued to exist. However, in contrast to the previous analysis, fewer factors appeared as significant in the resulting final models.

Need for further research or additional data

During the discussion of the report, WGPDMO stressed that there is still a lack of data in the ICES data banks (e.g., for contaminants in water, sediments and biota, but also for fish disease data covering other areas and species), which creates problems in the statistical analysis, particularly if the analysis is extended to cover larger regions in the ICES area. Since there is evidence that more relevant data are available in national data banks of ICES Member Countries, WGPDMO strongly emphasized the need to incorporate these data in the ICES data banks.

The ACME endorsed the WGPDMO view that the analysis of the ICES fish disease data should be continued and that a progress report should be presented and reviewed at the 2001 meetings of WGPDMO and ACME. For a more comprehensive use of available data, the disease prevalence data used should not be restricted to female dab, size group 20–24 cm, but should also involve other size groups and data for male fish. Furthermore, based on the availability of data, other geographical regions within the ICES area should be identified which can be included in the analysis.

Recommendation

ICES strongly encourages Member Countries to submit relevant new and historic data to the ICES data banks in order to enable a more comprehensive statistical analysis of data in relation to the spatial and temporal occurrence of fish diseases and the identification of their causes. Major gaps identified concern data on contaminants in sea water, sediments, and biota as well as data on the occurrence of diseases of wild fish outside the North Sea and of fish species other than common dab and European flounder.

References

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9.3 Update on the M74 Syndrome in Baltic Salmon and the Status of *Ichthyophonus* in Herring

Request

This is part of the continuing ICES work of updating the present knowledge on the causes of the M74 syndrome in Baltic salmon (*Salmo salar*) and progress in understanding the implications of relevant environmental factors (earlier, this was a request of the Helsinki Commission) and on the status of *Ichthyophonus hoferi* in herring (*Clupea harengus*).

Source of the information presented

The 2000 report of the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) and ACME deliberations.

Status/background information

The ACME reviewed the relevant sections of the WGPDMO report providing information on the present state of knowledge of the causes of the M74 syndrome in Baltic salmon and on the status of *Ichthyophonus hoferi* infections in herring.

M74

M74 continues to threaten Baltic salmon. Thiamine treatment of broodfish, eggs or fry is now routinely used in most hatcheries in Finland and Sweden to control fry mortalities. After two years of declining prevalences, there was a steep increase in 1999 offspring. A prognosis based on the thiamine contents of the eggs suggests that the prevalence will remain high in year 2000 offspring. Although progress was made, there was no significant breakthrough in 1999 in research focusing on the aetiology of M74.

The results obtained so far indicate a positive correlation between M74 and the increasing sprat population in the Baltic Sea, but this does not explain the aetiology of the disease. Investigations have shown that low thiamine contents in forage fish of Baltic salmon are not a cause of the disease, as the thiamine contents in herring and sprat are above the level necessary for salmon. Recent studies show that the agent(s) responsible for the disease appear(s) to be present in most parts of the Baltic Sea. The salmon populations with feeding runs to the main Baltic Sea as well as those migrating only to the Gulf of Bothnia and the Gulf of Finland have all developed the M74 syndrome. So far, there are no reports of M74 in the Latvian salmon populations, which have different patterns of feeding and spawning runs. Recent studies indicate that the gut microflora might be associated with the thiamine/thiaminase kinetics of fish.

Several experimental models for reproducing thiamine deficiency and M74 symptoms in fish, under laboratory conditions, have now been developed. These will facilitate research on the aetiology of the disease. Ongoing and recently initiated work is focusing more intently on lower levels in the food chain of Baltic fish, i.e., on the effects of the eutrophication and accompanying algal blooms, changes in plankton fauna, etc., as potential factors in the thiamine/carotenoid status of fish.

Ichthyophonus

Ichthyophonus hoferi infection continues to persist at a low prevalence in the European herring stocks examined, i.e., in Icelandic waters, the Kattegat, the northern North Sea, the Barents Sea, and the Norwegian Sea, without any indication of an epizootic.

There are no reports from the USA of obvious *Ichthyophonus* lesions from groundfish surveys in the Northwest Atlantic. Nor were any lesions found in Prince William Sound (Alaska) herring, in spite of a regular examination.

Need for further research or additional data

The aetiology behind M74 is still not clear and the disease syndrome remains a serious threat to wild salmon populations in the Baltic Sea. Therefore, it remains important that Member Countries continuously monitor salmonid populations for the occurrence of M74 or M74-like reproductive disorders. In light of the recent trend of seriously increasing M74 prevalence, the ACME emphasizes the urgent need for increased research efforts regarding the causes of this disease.

The persistence of *Ichthyophonus* in European herring stocks makes it necessary to continue the monitoring of the prevalence of this infection in herring stocks.

Recommendations

ICES recommends that Member Countries:

a) continuously monitor salmonid populations for the occurrence of reproductive disorders similar to the M74 syndrome;

b) continue to monitor the prevalence of *Ichthyophonus hoferi* infection in herring as a part of the fish stock assessment work.

9.4 Diseases and Parasites in Baltic Sea Fish

Request

Item 3 of the 2000 requests from the Helsinki Commission: to prepare Chapter 9 on “Marine fish migratory and freshwater species in the Baltic Sea Area” for the Fourth Periodic Assessment of the State of the Baltic Marine Environment, 1994–1998, comprising:

1) exploited species, 2) non-commercial species, 3) diseases and parasites, 4) effects of rearing upon wild populations, and 5) ecosystem effects of fishing activities.

Source of the information presented

The 2000 report of the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) and ACME deliberations.

Status/background information

The ACME reviewed a report on fish diseases and parasites in the Baltic Sea, prepared intersessionally for consideration at the 2000 WGPDMO meeting. The material presented was compiled by T. Lang (Germany) with input from other WGPDMO members from Baltic Sea countries and was intended as a contribution to the chapter on fish that ICES is preparing for the HELCOM Fourth Periodic Assessment.

The report represents an update of a chapter published in the HELCOM Third Periodic Assessment for the period 1989–1993 (HELCOM, 1996). A similar structure is being used for the new report, but it focuses on disease and parasite information for the period 1994–1998. The main fish species covered in the report are flounder, cod, herring, and salmon. However, other species with significant diseases or parasites are also considered. The report also contains current information on the impact of anthropogenic activities on the prevalence and geographical distribution of fish diseases and parasites and on the impact of fish diseases and parasites on Baltic fish stocks. A new section describing diseases in farmed marine fish in the Baltic has been added.

Since HELCOM indicated that the length of the chapter on Baltic fish stocks to be incorporated in the HELCOM Fourth Periodic Assessment has to be reduced as compared to the chapter included in the HELCOM Third Periodic Assessment, a shortened version of the fish disease report was prepared for submission to HELCOM (see Section 18.2, below). However, the ACME agreed to attach the full report on diseases of Baltic fish as Annex 6 to its report, since it contains detailed information that can usefully supplement the material supplied to HELCOM.

Reference

HELCOM. 1996. Third Periodic Assessment of the State of the Marine Environment of the Baltic Sea, 1989–1993: Background document. Baltic Sea Environment Proceedings, No. 64 B. 252 pp.

9.5 Other Fish Disease Issues

Request

This is part of continuing ICES work to consider new developments with regard to environmentally related fish disease issues.

Source of the information presented

The 2000 report of the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) and ACME deliberations.

Status/background information

In addition to the topics covered in other parts of Section 9, WGPDMO considered two other disease issues relevant to the work of ACME, as described below.

Distribution and effects of VHS-like virus in cultured and wild fish

In contrast to classic Viral Haemorrhagic Septicaemia (VHS), with its characteristic symptoms of disseminated haemorrhages, known from infections in rainbow trout (*Oncorhynchus mykiss*) in fresh water, the marine VHS-like virus infections occur with limited symptoms such as skin lesions or without gross lesions. Only turbot (*Scophthalmus maximus*) infected with marine VHS-like virus has revealed symptoms similar to the classic symptoms described from rainbow trout.

In the Pacific area off North America, marine VHS-like virus was observed for the first time in ascending chinook salmon (*O. tshawytscha*) and in coho salmon (*O. kisutch*) in 1988 and since then it has been isolated from six Pacific marine fish species.

In Europe, the awareness of VHS in marine fish species was raised in association with an outbreak of VHS in farmed turbot in Scotland in 1994. Prior to this observation, VHS virus had been associated with outbreaks in rainbow trout in France (1980) and Denmark (1982) and in turbot in Germany (1991), but the source of these infections was regarded as being of freshwater origin. In the marine environment, VHS virus had been isolated from cod in 1979 and 1993 and from haddock in 1993, but these findings were regarded as being obscure or as a product of contamination.

Intensive monitoring on the geographical distribution and the host range has been initiated in many European countries. So far, these investigations have shown that marine VHS-like virus occurs in the British Channel, northern North Sea, North Atlantic north of Scotland, Skagerrak, Kattegat, and the Baltic Sea, with the highest prevalence in the Baltic Sea. So far, marine VHS-like viruses have been isolated from fourteen marine fish species in Northern European waters.

Mass mortalities in several marine fish species associated with marine VHS-like virus have been reported since 1998 from the North American Pacific area.

The marine VHS-like isolates have shown low pathogenicity to salmonids where tested in Europe and North America. This has been in contrast to isolates from fresh water.

Genetic analyses of marine VHS-like viruses have so far revealed four different groups, namely a Pacific strain, a North Atlantic Scottish strain and two strains from the Baltic/Kattegat, one of which appears to be closely related to the freshwater isolates.

Need for further research or additional data

The ACME recognized that marine VHS-like virus may constitute a potential risk for the culture of marine fish species because of the wide host range so far detected. There is also evidence accumulating that, in some circumstances, marine VHS-like virus infection may be associated with mass mortalities in wild marine fish populations, e.g., Pacific herring (*Clupea harengus pallasi*).

During discussion, it was recognized that there are important gaps in knowledge, particularly in the two areas listed below.

The infectivity and pathogenicity characteristics of the different marine isolates of VHS-like virus are emerging as particularly important features, useful in conjunction with genetic information, to distinguish different types or strains within this now demonstrably heterogeneous virus group. However, to date there has only been limited cross-infectivity trials using different isolates and different types of non-salmonid marine fish species.

Considerable evidence is accumulating that different viral types are being listed under the same name, Viral Haemorrhagic Septicaemia. As previously noted by WGPDMO, the use of VHS as a name for marine rhabdoviruses is causing confusion. However, the justification for adopting different names for different strains is still doubtful, because of the current inability to differentiate the different isolates, particularly fresh water and marine. It is considered that no re-naming of these agents should take place until differential detection and diagnostic techniques have been developed, standardized, and validated.

Proposal for incorporating parasitological studies into existing fish disease monitoring programmes

There are a number of examples from the literature where a link between changes in the occurrence of parasites of wild marine fish species and environmental change, including anthropogenic effects, could be demonstrated (e.g., effects of acidification, sewage sludge dumping, oil contamination, and eutrophication). Therefore, parasites are considered to have potential to be used as biological indicators in environmental monitoring programmes.

The identification of cause-effect relationships related to changes in temporal and spatial distribution patterns of parasites is complicated because a wide range of biotic and abiotic factors, including the distribution and abundance of intermediate hosts, affect the occurrence of parasites. However, even if it is not immediately possible to identify causes of changes, parasites may be useful as “alarm bells”, initiating further more in-depth studies.

Two basic strategies can be applied for studies using parasites as indicators of environmental change: a) to investigate alterations in the entire parasitofauna of a certain fish species, and b) to focus on specific parasites or groups of parasites known to be affected by changes in environmental parameters. A disadvantage of the first strategy in terms of its use for monitoring purposes is that it is labour-intensive and that only a few fish specimens can be examined per sample. However, a comprehensive analysis is considered useful as it provides relevant background information for selecting suitable parasite/host combinations. The application of the second strategy implies a proper knowledge of the biology of the parasites and their host, including information on their responsiveness to environmental change.

It was emphasized that the selection of parasites to be used as indicators for monitoring purposes depends on the objectives of the monitoring. For local short-term effects (e.g., in coastal areas), different species may be suitable than for long-term effects in larger areas (including offshore areas).

Need for further research or additional data

The ACME considered parasites of wild marine fish to be of potential value as biological indicators of environmental change. However, before they can be recommended for monitoring purposes, a number of requirements have to be met, e.g., with regard to the following issues:

- the objectives of monitoring programmes involving parasites as biological indicators have to be clearly defined;

- parasites and their host species used for monitoring purposes have to be carefully selected based on knowledge of their life cycles and their responsiveness to environmental change;
- there is a general need for more information on the biology of parasites and their hosts as well as their interactions;
- as a basis for a more comprehensive evaluation of the use of parasites as indicators, existing data sets on parasites of wild fish must be compiled and analysed in order to obtain a better impression of the extent of local and temporal variation in their occurrence and the impact of environmental factors.

9.6 New Techniques in Pathology for the Detection of Endocrine Disrupting Chemicals in Marine and Estuarine Organisms

Request

This is part of continuing ICES work to develop techniques to detect biological effects of contaminants in the marine environment.

Source of the information presented

The 2000 report of the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) and ACME deliberations.

Status/background information

The topic of endocrine disruption in marine and estuarine species continues to be of high priority and there remains the need to develop robust techniques for the detection of endocrine disrupting chemicals and their effects in a variety of organisms. Although most attention has been given to the effects of oestrogen mimics, there is increasing effort toward the development of methods for testing for androgens and thyroid-acting substances.

Almost all research has been devoted to responses in fish hosts and (within ICES Member Countries), in particular, in the European flounder (*Platichthys flesus*). This species has been shown to be susceptible to endocrine disruption and exhibits elevated levels of vitellogenin (VTG) in certain estuaries. In addition, the induction of intersex condition in male flounder has been shown to occur in contaminated estuaries (Matthiessen *et al.,* 1998; Allen *et al*., 1999a, 1999b). VTG induction in male fish is generally regarded as a very sensitive biomarker of oestrogen induction and is now in use in several laboratories. However, it is apparent that some fish species are relatively insensitive to VTG induction.

The histological endpoint is valuable in demonstrating the induction of intersex in otherwise male fish, but the relationship between the physical condition and elevated VTG levels still needs to be clarified since it is not clear that intersex fish also show elevated VTG levels. However, the use of histology requires a high level of technical skill and access to an experienced pathologist familiar with the species under study. Additional endpoints currently in use are effects on gonadosomatic index and effects on gross morphology, particularly secondary sexual characteristics. These are currently being studied in goby species. Higher level effects on sex ratios and fecundity also need to be assessed, however, measures of effects on reproductive success will involve significant method development. There is scope for measuring the reproductive success of individual viviparous blenny (*Zoarces viviparus*) and preliminary data from the UK show that this species is susceptible to the induction of VTG and intersex conditions.

Similar techniques are also being applied to investigate effects on various crustaceans, including *Carcinus, Chaetogammarus* and *Crangon* species, with emphasis on vitellin measurements and effects on breeding success. However, recent data indicate that vitellin induction is not an appropriate indicator of oestrogenic exposure, since no induction could be detected in male crustaceans, either in experimental or in field studies. Histological endpoints are also likely to be of value, but until sufficient baseline data have been accumulated, it is difficult to be clear on the specific histological parameters that may be used. New approaches using immuno-histochemical methods are also beginning to be used for marine species. These methods will be valuable in enabling detection and localization of VTG or vitellin in various fish and crustacean species. The development of such techniques will prove essential in efforts to understand physiological and physical changes associated with exposure to endocrine disruptors and could provide a valuable tool for detection of early effects.

A biomarker of androgen exposure in sticklebacks (*Gasterosteus aculeatus)* has been developed based on the protein spiggin, which is produced in male sticklebacks and secreted from the kidneys and used in nest-building activities. A histological method measuring renal tubule cell hypertrophy has also been developed. It is possible that similar techniques could be used in other nest-building marine fish that utilize spiggin or a similar protein (Katsiadaki *et al*., 2000).

Mollusc species have not attracted much attention as susceptible species to endocrine disruption, apart from a large volume of work that has been done on TBT-related imposex and intersex in gastropods. Although it is clear that molluscs form a very important component in the marine ecosystem, the current lack of methods to detect endocrine disruption in this group imposes a barrier to their use.

Need for further research or additional data

The ACME endorsed the view of WGPDMO that, although there exist some pathological tools to be used in biological effects studies of endocrine disrupting chemicals, research is still needed to refine these methods and to develop new techniques before histopathology can be fully applied in such studies.

References

Allen, Y., Scott, A.P., Mathiessen, P., Haworth, S., Thain, J.E., and Feist, S. 1999a. Survey of estrogenic activity in United Kingdom estuaries and coastal waters and its effect on gonadal development of the flounder *Platichthys flesus*. Environmental Toxicology and Chemistry, 18: 1791–1800.

Allen, Y., Scott, A.P., Haworth, S., Feist, S., Thain, J.E., Hutchinson, J.H., and Matthiessen, P. 1999b. The extent of oestrogenic contamination in the UK marine environment—further surveys of flounder. The Science of the Total Environment, 233: 5–20.

Katsiadaki, I., Scott, A.P., and Matthiessen, P. 2000. The use of the three-spined stickleback as a potential biomarker for androgenic xenobiotics. *In* Proceedings of the 6th International Symposium on the Reproductive Physiology of Fish, Bergen, Norway, 4–9 July 1999, pp. 359–361. Ed. by B. Norberg, O. Kjesbu, G.L. Taranger, E. Anderson, and S.O. Stefansson.

Matthiessen, P., Allen, Y.T., Allchin, C.R., Feist, S.W., Kirby, M.F., Law, R.J., Scott, A.P., Thain, J.E., and Thomas, K.V. 1998. Oestrogenic endocrine disruption in flounder (*Platichthys flesus*) from United Kingdom estuarine and marine waters. CEFAS Science Series Technical Report 107, Lowestoft, UK. 48 pp.

10 INTRODUCTIONS AND TRANSFERS OF MARINE ORGANISMS

10.1 Current Status of Fish, Shellfish, Algal, and Other Introductions in and between ICES Member Countries

Request

ICES Member Countries may request ICES to review proposed introductions and transfers of marine organisms for mariculture purposes. These proposals receive in-depth review by the Working Group on Introductions and Transfers of Marine Organisms (WGITMO), with final review by the ACME. WGITMO also keeps under review the progress of such introductions and reports the outcome to the ACME.

No new requests for review of proposed introductions were received in 2000, but the status of on-going introductions and transfers was reviewed.

Source of the information presented

The 2000 report of the Working Group on Introductions and Transfers of Marine Organisms (WGITMO) and ACME deliberations.

Status/background information

The ACME reviewed the WGITMO report and agreed to present the information contained in the following sections.

10.1.1 Status of existing controlled introductions

Japanese seaweed (Nori) *Porphyra yezoensis* in the USA

The ACME noted that the reports on biomonitoring of *Porphyra yezoensis* and on a review of the performance of a number of different test sites over the period 1992–1999 had been finalized.

The conclusions of the independent monitoring programme (Watson *et al*., 2000) were as follows:

1) *P. yezoensis* plants are present but uncommon on the shoreline adjacent to the farm sites during the growing season;
2) local *Porphyra* species out-recruit *P. yezoensis* on netting substrates;
3) there is no evidence to date that *P. yezoensis* will over-winter in Cobscook Bay and replace local *Porphyra* species.

Despite extensive trials in a number of sites, attempts to cultivate “Nori” *P. yezoensis* commercially over a period of nine years have not been successful due to a combination of regulatory and extreme hydrographic factors. Winter survival was poor. Presently there are no further plans for commercial cultivation of *Porphyra* species in general and *P. yezoensis* in particular.

10.1.2 Status of accidental introductions

The ACME noted that accidental introductions and transfers of non-native species continue to be documented in many ICES Member Countries; these include the following:

1) The Japanese carnivorous whelk (snail) *Rapana venosa* continues to be observed in Chesapeake Bay, USA. The current number of specimens is stated to be over 800. The distribution centre remains the same as in the previous two years, with little expansion. Catches of “smaller” individuals in the spring of 2000 may be evidence of breeding; there are also limited local collections of egg masses.

   Under laboratory conditions, broodstocks were demonstrated to produce 530 egg masses between May and August 1999. Cultured juveniles have been observed to prey on a range of local macrofauna including bryozoa, barnacles, mussels, oysters (*Crassostrea virginica*), and soft clams (*Mya arenaria*).

   Adults of this species have been found in the Bay of Quiberon in Brittany, France, but so far no juveniles have been found and there is no evidence yet that a local population has been established. Genetic comparisons with US material may provide evidence of origin.

2) The green crab, *Carcinus maenas*, has now been found on the west coast of Vancouver Island, having originally been reported in Prince Edward Island in 1998. There is considerable concern about its expansion into prime shellfish farming and/or harvesting areas in the Gulf of St. Lawrence.
3) Mortalities of the abalone *Haliotis tuberculata* occurred in Brittany from a *Vibrio* (*V. courtier*) known to induce mortality in *H. discus hannai.* These mortalites are associated with illegal imports of *Haliotis* from Ireland. It is believed that they carried the *Vibrio,* which then expressed itself in the warmer temperatures of southern Brittany.
4) The zebra mussel, *Dreissena polymorpha*, has increased in density in the Shannon, Erne, and Boyle river systems in Ireland, and further expansion will occur with planned extension of navigable waters. An awareness campaign has been conducted with national angling groups to prevent the spread by boat movements to important but vulnerable fishing lakes.
5) The cladoceran zooplankton species *Cercopagis pengoi* was very abundant in both Polish and

Estonian waters in the Baltic Sea, coinciding with the highest summer water temperatures. Studies in the Baltic Sea indicate that the spatio-temporal dynamics of populations of this species are strongly dependent on climatic conditions such as water temperatures and the stability of the water column. Populations are more abundant in sheltered areas and the animals are not found in environments characterized by strong currents. In Finland, this species extended its range up to 63 °N. *Cercopagis pengoi* is the latest exotic crustacean to invade the North American Great Lakes. Canadian scientists first identified this predatory cladoceran in early August of 1998. *Cercopagis* is indigenous to the Caspian, Azov, and Aral Seas, and was reported to have invaded the Baltic Sea in 1992.

6) The spread of a seaweed identified as *Codium fragile tomentosoides* in Atlantic Canadian waters has been observed since 1996. Authorities continue to monitor the *C. fragile* populations in Enmore Bay (Northumberland Strait side of the Island) and Malpeque Bay (Gulf of St. Lawrence side) and experiment with various treatment immersion trials in an attempt to control the seaweed's spread with shellfish movements. The plant has not been identified outside of these areas, other than the one isolated finding on mussel lines in Tracadie Bay (also on the Gulf of St. Lawrence side). Results indicate that the plant is very difficult to kill and that it takes a long period of time to assess treatment efficacy. Trials to control the spread of the seaweed *Codium fragile* on oyster spat by treatment with saturated brine, lime, air drying or a combination of all three were unsuccessful.

7) The polychaete *Marenzelleria viridis*, known in Swedish waters since 1990, continues to increase somewhat in the northern part of the Bothnian Sea, now occurring as far north as on the western side of Holmoarna as well as east of these islands and continues to spread further south in the Bothnian Sea. In the Askö-Landsort area, northern Baltic Proper, the species was found in 1999 for the first time both west of Landsort and east of Askö, the size and number of the individuals on one site indicating it had been there for a few years. The species is still not present in high abundance in the southern Baltic. There are no reports of *Marenzelleria* from the west coast of Sweden.

Seaweeds

Caulerpa taxifolia

Since the first sighting of *Caulerpa taxifolia* on the French Mediterranean seaside, this species has extended its distribution considerably (4600 ha. in 1998), colonizing areas in Croatia, Monaco, Italy, and Spain. An additional exotic species *C. racemosa* is presently observed in the western Mediterranean Sea and was recently reported in Genoa (Italy) and Marseille (France). In 1998, several cruises were carried out to assess the present distribution of *Caulerpa taxifolia* (4600 ha.). First sightings of *Caulerpa* within dense *Posidonia oceanica* seagrass were observed in 1998. The depth distribution showed *C. taxifolia* pieces down to 108 m, while the sampling was carried out to a depth of 182 m. The direct predation on *Caulerpa taxifolia* by the mollusc *Lobiger serradifalci* was confirmed. Several trials of destruction by using electrodes are currently under investigation. The second European research programme LIFE aims to specify management tools to limit further expansion in the Mediterranean Sea. Although in progress, a report entitled "Contrôle de l'expansion de *Caulerpa taxifolia* en Méditerranée" has been published.

Finfish

The round goby, *Neogobius melanostomus*, has recently spread to the eastern coast of Poland. In 1999 the first specimen was found in the Vistula Lagoon. This fish species, which lived in the mouth of the Vistula River, recently has moved up river. A few specimens were found on the coast of Rügen Island in the southwestern Baltic. New settlements of round goby in the southwestern Baltic are probably the result of its introduction by means of ballast water because, at the moment, there are no other records from southwestern Baltic waters. If this theory is correct, it means that anthropogenic spreading of the round goby has no limitations, except habitat conditions. The round goby has become a stable element in the trophic chain in the Gulf of Gdansk. Cod frequently eat it during feeding in coastal areas. Birds (cormorants) also feed on it.

There are still no reports of *Neogobius melanostomus* from Swedish coastal waters, despite its common occurrence in the Gulf of Gdansk.

Specimens of round goby, *Neogobius melanostomus*, have been transferred by inter-lake shipping to Hamilton harbour and the east end of Lake Ontario near Kingston in Canada. One specimen was also captured in the St. Lawrence River near Quebec City.

Recommendations

ICES notes that it is not clear that all of these accidental introductions are of major ecological significance and recommends that a report be prepared on the current status of previously reported introductions in order to assess their long-term implications.

Reference

Watson, K., Levine, I., and Cheney, D. 2000. Biomonitoring of an aquacultured introduced seaweed, *Porphyra yezoensis* (Rhodophyta, Bangiophycidae) in Cobscook Bay, Maine, USA. Proceedings of the First National Conference on Marine Bioinvasions, MIT, Cambridge, MA.

10.2 Progress in Ballast Water Research and Management

Request

This is part of the continuing ICES work to keep under review new information concerning ballast water research and management.

Source of the information presented

The 2000 reports of the ICES/IOC/IMO Study Group on Ballast and Other Ship Vectors (SGBOSV) and the Working Group on Introductions and Transfers of Marine Organisms (WGITMO), and ACME deliberations.

Status/background information

EU Concerted Action on "Testing monitoring systems for risk assessment of harmful introductions by ships to European waters"

One key objective of the EU Concerted Action was to test monitoring systems for sampling ballast water. Two major intercalibration workshops compared sampling techniques. The two workshops provided results that allow for better comparisons of ship sampling studies around the world. A second key issue was to gain more insight on species composition in ballast water during ship voyages, and this was achieved by ocean-going workshops, which provided data collected during five such workshops.

Public awareness was achieved by preparing a video, a leaflet, flyers, press releases, newsletter articles of International Aquatic Societies, an internet homepage, and several posters. A book on case histories, listing species previously introduced to European waters, was prepared for harbour and regulatory authorities.

National reports

Ten ICES Member Countries and New Zealand submitted activity reports on ongoing research activities to detect the presence of new species of organisms present in ship's ballast water and sediments. Projects were designed with various purposes in mind, as follows:

- assessing risks for the introduction of exotic species through ballast water in areas of foreign maritime traffic;
- providing a scientific background to better manage these risks;
- testing potential ballast water treatment methods as an alternative to offshore ballast water exchanges to reduce risks of ballast water mediated introductions;
- establishing monitoring programmes to detect the presence of invasive species in ballast water and sediments in various locations;
- assessing the dilution efficiency of mid-ocean ballast water exchange as well as the efficacy of such exchanges at expelling organisms resident in the ballast water before the exchange;
- assessing the survival of planktonic organisms while in ballast water.

Directory of dispersal vectors of exotic species

The ACME noted that the Working Group on Introductions and Transfers of Marine Organisms (WGITMO) discussed the range of vectors involved in introductions and transfers of marine organisms. The compilation of information for a new *ICES Cooperative Research Report* containing a "Directory of Dispersal Vectors" is nearing completion by WGITMO.

Databases on introduced species

WGITMO reported that many databases on introduced and transferred species have been developed or are in the process of development, and that many are specific to geographical regions. Such databases are valuable sources of information to scientists and other groups (e.g., HELCOM) in ICES Member Countries as well as around the world. The availability of information on websites is increasing exponentially. Databases as a source of information for assessing risks from ballast water introductions represent only one example of the value of these regional databases.

The ACME noted, however, that there is a need to review databases on introduced species that have been developed (or those in development), on a regional basis, to improve communication and the dissemination of information within and between ICES Member Countries, and to inform other groups, such as HELCOM, where information on introduced species can be found, since ICES does not presently maintain such a database.

Need for further research or additional data

The ACME agreed that it is necessary to prepare a detailed overview of the status of biological and ecological research on ballast water and sediment, and on the development of ballast water control and management technologies. It is also necessary to evaluate the relationship between ballast water movement and the invasion of exotic marine organisms, including updates on the latest ballast-mediated invasions globally, particularly relative to those species that are now invasive in other regions of the world and that are ballast-transportable, but have not yet arrived in ICES Member Countries.

11 ESTIMATION OF ANNUAL AMOUNT OF DISCARDS AND FISH OFFAL IN THE BALTIC SEA AND POTENTIAL EFFECTS

Request

Item 5 of the 2000 requests from the Helsinki Commission: to provide information on the possible impact on the Baltic marine environment of dumping of fish remnants, especially regarding:

- size of disposal of fish remnants from fish-processing units,
- amounts of disposal of undersized fish by fishermen,
- possible secondary effects caused by dumping of fish remnants.

Source of the information presented

The 2000 reports of the Study Group on Estimation of the Annual Amount of Discards and Fish Offal in the Baltic Sea (SGDIB), the Working Group on Seabird Ecology (WGSE), and the Benthos Ecology Working Group (BEWG); the 1999 report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO); and ACME deliberations.

Status/background information

A Study Group on Estimation of the Annual Amount of Discards and Fish Offal in the Baltic Sea (SGDIB) was created in 1999 to respond to this request from the Helsinki Commission. SGDIB has fulfilled its task by elaborating the available data on national landing statistics (including discards) from 1998 and the results of the ongoing EU project "International Baltic Sea Project II".

Discards

Fishing activities result in the capture of non-target species and undersized individuals of target species. A proportion of this by-catch is discarded dying or dead because it is illegal to land it or because there is little or no economic gain associated with sorting or retaining it in relation to other catches. In fisheries that are managed using quotas, target fish above the minimum legal landing size may be rejected in favour of larger, more valuable specimens.

Monitoring programmes for collecting information on discards in various fisheries are expensive. Very limited discard monitoring was conducted prior to 1995, when the European Commission decided to co-finance discard monitoring schemes in the Baltic Sea. Five countries around the Baltic Sea participated in the first scheme, and all countries with the exception of Lithuania participated in the next two schemes. Although data collected in the framework of this project were available for the years 1995–1999, since this was the first attempt to estimate the total amounts of discards in the Baltic Sea, time constraints only permitted estimates to be made for one year. As the most data were available for 1998, the estimates were made for this year.

Findings on the amounts of fish discarded in the Baltic Sea in 1998 in the different Sub-divisions (see map in Figure 11.1) are given in Tables 11.1 and 11.2.

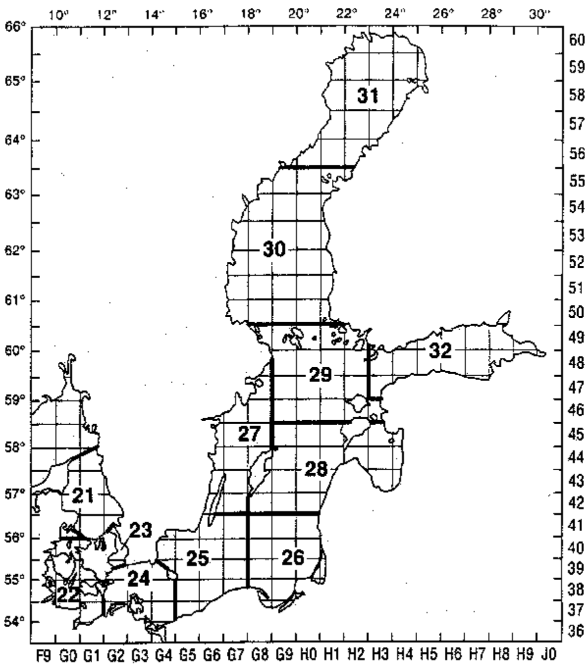
Table 11.1. Estimated amount of landings of fish (tonnes) and discards in the Baltic Sea in 1998, and the overall discard percentage by Sub-division.

Sub-division	Landings (tonnes)	Discards (tonnes)	Overall discard percentage
22	37 567	4 432	10.6 %
24	44 703	2 058	4.4 %
25	201 929	3 598	1.8 %
26	123 778	786	0.6 %
27	120 603	20	0.0
28	101 636	104	0.0
29	57 397	0.4	0.0
30	53 118	0.5	0.0
31	4 817	2.5	0.0
32	15 543	0.9	0.0
Total Baltic Sea	**761 091**	**11 003**	**1.4 %**

Total landings of fish in 1998 were estimated at 761 091 tonnes, while discards were estimated at 11 003 tonnes. This gives a total catch of 772 094 tonnes and an overall discard rate equal to 1.4 % by weight. This figure shows that the amount of discards is close to negligible. It is important to note, however, that most of these discards occur in the southwestern Baltic and the western part of the Bornholm Deep (Sub-divisions 22 and 24), where they amounted to 10.6 % and 4.4 %, respectively. These significant discards most probably came from undersized cod (< 35 cm) (in 1997 an abundant year class of cod developed). If industrial fisheries, where no discarding normally is practiced, are excluded, the overall discard rate in the remaining fisheries is 3.8% by weight. Catches in the other parts of the Baltic area were very clean: zero or close to zero discards.

The largest amount of discards relates to cod (6 573 t), followed by flounder (2 089 t), sprat (910 t), plaice (515 t), and dab (390 t). It is also worth noting that the eastern cod stock in the Baltic Sea is severely depleted at the present time and if cod stocks improve, discards will likely be larger.

Figure 11.1. Map of the ICES Fisheries Sub-divisions of the Sound, Belt Sea, and Baltic Sea used in fishery assessment.



Discards related to fish species indicate that there were more than thirty different fish species discarded in the Baltic (discarding of herring is prohibited). This large number may possibly result from non-residential, visiting fish (e.g., starry ray, Norway pout, mackerel).

Possible discards from trap-nets (pound nets, fyke nets, etc.) have not been considered because no realistic landings statistics are available for these gears; vessels operating these gears typically are too small (normally below 10 m) to be obliged to fill in logbooks. For fisheries registered as landing fish but where no discard data are available, the discard rate is defined as zero if, by experience, it is evident that no or a negligible amount of discarding occurs. Furthermore, no discard data are available for fisheries directed toward salmon and it is assumed that the amount of discards in these fisheries is negligible.

Sweden only includes cod in its database concerning discards. Therefore, Swedish discards of species other than cod are not included in the estimate. As such, the estimate of total discards in the Baltic Sea is probably underestimated by roughly 800–900 tonnes, based on the Danish ratio between the discards of cod and the sum of other species discarded.

Fish offal

Some of the fish caught in the Baltic Sea are gutted at sea and some parts of the guts or all of the guts are disposed in the sea. For some species, parts of the guts such as the liver and female gonads are separated and landed. In the Baltic, the number of commercially important species is limited and the most important species in tonnage, herring and sprat, are not gutted at sea.

Table 11.2. Estimated amounts of fish (tonnes) discarded in the Baltic Sea in 1998 by species. Species for which less than 5 kg have been discarded per year are excluded.

Species	Discards (tonnes)	Species	Discards (tonnes)
Starry ray	26.3	Scorpion fish	217
Herring	14.4	Armed gurnard	0.05
Sprat	910	Lumpfish	19.6
Sea trout	7.1	Perch	0.66
Smelt	10.1	Black tail	2.3
Rutilus rutilus	0.54	Lesser sandeel	9.4
Cod	6 573	Greater sandeel	1.9
Saithe	0.12	Mackerel	0.14
Haddock	2.1	Turbot	68
Four-bearded rockling	42.3	Brill	8.4
Poor cod	0.05	Long rough dab	21.1
Norway pout	0.13	Dab	390
Whiting	51.5	Lemon sole	0.36
Eelpout	22.2	Flounder	2 089
Grey gurnard	0.08	Plaice	515
Four-horn sculpin	0.08		

Table 11.3. Estimated total amount of offal (in tonnes) from cod for 1998 by quarter of the year and Sub-division.

Sub-division	Quarter 1	Quarter 2	Quarter 3	Quarter 4	Total
22	1 300	409	288	752	2 792
24	818	551	457	974	2 799
25	1 556	2 516	957	2 064	7 093
26	1 208	1 995	523	1 397	5 124
27	101	84	10	31	225
28	51	58	57	205	371
29	0	6	0	0	7
30	0	1	0	0	1
31	0	0	0	0	0
32	1	1	0	0	2
Total	5 035	5 621	2 292	5 423	18 414

Table 11.4. Estimated amount of offal (in tonnes) from the gutting of cod that is discarded in the Baltic Sea, by quarter of the year and Sub-division.

Sub-division	Quarter 1	Quarter 2	Quarter 3	Quarter 4	Total
22	1 057	299	257	652	2 265
24	665	402	407	845	2 320
25	1 265	1 837	853	1 792	5 747
26	982	1 457	467	1 213	4 118
27	82	61	9	27	179
28	42	42	51	178	313
29	0	4	0	0	5
30	0	1	0	0	1
31	0	0	0	0	0
32	1	1	0	0	2
Total	4 094	4 104	2 044	4 707	14 950

Conversion factors for gutted to live weight were developed for the species of fish gutted at sea: cod, salmon, sea trout, flounder, turbot, plaice, and dab. Herring, sprat, whitefish, perch, pike, and pikeperch are not gutted at sea; therefore, no conversion factor was necessary for these species. Taking the total landings and the mean conversion factors into account, only the gutting of cod contributes considerably to the total amount of fish offal. Offal or guts of cod comprise up to 19 % of total fish weight (on average per year), ranging from 15 % to 26 %. Only calculations on cod have been made here, as the amount of fish offal from gutting of other species is insignificant. The total amount of cod offal is given in Table 11.3. However, not all of this is dumped, as the gonads of mature female cod and good quality cod liver are collected and sold. The estimated amounts of offal actually dumped are shown in Table 11.4.

As shown in Table 11.5, the total dumping of offal and discards is the greatest in Sub-division 25 estimated at 9345 t, followed by Sub-division 22 at 6697 t. Quantities in Sub-divisions 24 and 26 are estimated at 4378 t and 4904 t, respectively. In terms of species, the discarding of cod is by far the largest amount, estimated at 6573 t per year and the offal dumped is also mainly associated with cod.

Table 11.5. Overall estimated amounts of offal and discards (in tonnes) dumped in the Baltic Sea in 1998 by Sub-division.

Sub-division	Total Offal	Discards	Overall amounts
22	2 265	4 432	6 697
24	2 320	2 058	4 378
25	5 747	3 598	9 345
26	4 118	786	4 904
27	179	20	199
28	313	104	417
29	5	0.4	5.4
30	1	0.5	1.5
31	0	2.5	2.5
32	2	1.0	3
Total	**14 950**	**11 002.4**	**25 952.4**

On an overall Baltic Sea basis, the amounts of offal discharged are roughly equal during Quarters 1, 2, and 4, but approximately half this amount in Quarter 3, given that most discards are associated with the cod fishery.

There are also other factors that can have an influence on the amount of fish offal in the Baltic area, including the influence of marine mammals, such as seal damage to salmon in the trap-net fishery. In recent years, catch losses have increased particularly in Sub-divisions 30–32. In coastal trap-net fisheries in the Gulf of Bothnia, the losses in catch due to seal interactions with the fishery exceed 200 tonnes, but the effect on the ecosystem is unknown.

Potential environmental effects

Due to a lack of research, the environmental effects of discharged fish remnants are largely unknown, but are likely to include effects on scavengers, and decomposition processes. Some estimates can be drawn from data from other seas, mainly the North Sea; however, due to different conditions in the North Sea, the significance of these comparisons is limited.

Studies in the North Sea indicate that considerable amounts of fish offal are consumed by seabirds (Hunt and Furness, 1996). Thus, the Working Group on Seabird Ecology (WGSE) was requested to estimate the proportion of fish offal and discards, based on the information from SGDIB, that might be consumed by seabirds in the Baltic Sea.

Based on its estimates (see Annex 7 for the details), WGSE concluded that, although there has been only a very limited amount of work on the consumption of discards by seabirds in the Baltic Sea, it is evident that herring gulls consume a high proportion of the fish offal and discard production in this area. A few discards are too large for gulls to swallow (predominantly cod over 27 cm), but there is evidence of gulls selecting roundfish discards and not taking some flatfish discards. The data suggest that gulls consume considerably more than half of the discards and almost all of the offal discharged by Baltic fisheries. In addition, it appears that herring gull distribution in the Baltic in winter is determined to a considerable extent by the local distribution of discarding fishing vessels, which are concentrated in the southwestern Baltic. Whether the provision of discards and offal from Baltic fisheries has affected the population trends of gulls in the Baltic is not known.

Overall estimates of the amounts of discards and offal consumed by seabirds in the Baltic Sea are given in Table 11.6. Even accounting for possible overestimation, it is clear that very large proportions of the fish offal and discards dumped into the Baltic Sea are consumed by seabirds.

The fish offal and discards that are not consumed by seabirds will be available for consumption by mid-water scavengers and benthic predators. Few studies have been made regarding the consumption of discarded material in the mid-water, probably owing to the sampling difficulties associated with these types of studies (Britton and Morton, 1994).

Table 11.6. Quantities of discards and offal produced annually in the Baltic Sea and their consumption by seabirds. See Annex 7 for details of the calculations.

Species	Amount (tonnes)	Amount consumed (tonnes)	Proportion consumed
Cod	6 573	5 587	85 %
Flounder	2 089	251	12 %
Sprat	910	737	81 %
Plaice	515	62	12 %
Dab	390	47	12 %
Scorpion fish	217	109	50 %
Turbot	68	8	12 %
Whiting	52	46	88 %
Other fish	188	94	50 %
Total discards	**11 002**	**6 941**	**63 %**
Offal	**14 950**	**14 502**	**97 %**

With regard to benthic predators, the Benthos Ecology Working Group (BEWG) was requested to consider potential effects of the discharge of fish offal and discards on benthos in the areas affected in the Baltic Sea. BEWG reported that there is a lack of knowledge on effects of these discharges on benthic community structure and biomass. However, based on deliberations of BEWG, the following topics should be taken into consideration:

- Which potential scavengers exist in each area and depth zone (e.g., decapods, *Saduria*, *Asterias,* polychaetes, gammarids, isopods, fishes, sea mammals)?
- What is their depth distribution?
- Feeding guilds of benthic species (including consumption rates) need to be investigated.
- Video inspections for discarded material should be conducted on frequently used fishing grounds with significant production of discards.
- Aerobic and anaerobic decomposition processes (costs versus benefits for the ecosystem, O_2, and nutrient recycling) should be reviewed.

Finally, the issue of the oxygen consumption required for the decomposition of fish offal in the sea was considered. The Working Group on Ecosystem Effects of Fishing Activities (WGECO) was asked to evaluate possible effects of offal discards in the Baltic. One aspect addressed was the question of the effects of offal on the oxygen deficiency in bottom waters. To analyse this, WGECO concentrated on the Baltic Proper where most of the fishery takes place and, 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 clearly shows that the oxygen requirement from discarded offal is insignificant (Table 11.7). The oxygen consumption owing to decomposition of discarded offal is < < 1 % of the consumption owing to decomposition of sedimented phytoplankton, when considered at the scale of the whole Baltic Sea.

Table 11.7. Relative contributions of primary production, sedimentation, and discarded offal to the carbon load of the Baltic Sea (most values from Elmgren, 1984).

Process	**g C m^{-2} annually**
Baltic Proper primary production	170
Sedimentation	57
Discarded offal[1]	0.028
Contribution by offal to total oxygen consumption	**0.05 %**

[1]High values from the early 1980s (ICES, 1997): 58 000 t carbon as proportion of wet weight: 0.1 surface area of the Baltic Proper: 2.1×10^{11} m^2.

To estimate possible secondary effects from the dumping of fish remnants, it is important to know not only the amount of fish remnants but also the location of discharges and the method of dumping. Although the contribution to total oxygen consumption is insignificant when considered at the scale of the whole Baltic Sea, massive discharges of fish remnants in a small area will obviously have negative effects on local oxygen conditions. The effect can be deliberately diminished if fish remnants are dumped over a larger area. Effects of dumping will also depend on actual environmental conditions on the bottom; since the Baltic Sea has large areas depleted of oxygen, it is important to avoid dumping fish remnants in these areas.

Need for further research or additional data

The ACME agreed that more efforts on the estimation of discards in the Baltic Sea should be undertaken by the Baltic fishery.

The discard calculations reported above were based on 2322 stations (hauls/set) sampled by observers on board commercial vessels. The number of stations sampled is estimated to constitute less than 1 % of the total fishing stations occupied by the commercial fleet. The relatively small coverage means that the sampling result is very dependent on the ability of the national sampling schemes to reflect all major fisheries by fleet, season, and fishing pattern in a balanced manner that reflects the total fishery. Not all countries have the capability to adjust the sampling scheme continuously to guarantee that the sampling fulfills this balanced approach. Therefore, the estimates are expected to be associated with significant variances and maybe some bias. The estimates should therefore be taken with some caution. Nevertheless, it is believed that the estimate of total discards is realistic and precise enough to imply the amount of discards in the Baltic Sea.

The ACME is aware that all of the results provided by SGDIB are based on the year 1998. Keeping in mind that the abundance of yearly classes of different fish species as well as fish catches can vary significantly, there is a need to include information and/or data from other years to provide a better estimation of the annual amount of discards and fish offal in the Baltic Sea. Therefore, the ACME recommends that, if information from additional years is available, SGDIB should continue its work to elaborate a better picture of the discharge of fish remnants to the Baltic Sea.

The ACME also encourages benthos ecology scientists to undertake research programmes on the evaluation of possible secondary effects on benthos in the Baltic Sea from the dumping of fish remnants. This work should lead to a more quantitative assessment of the effects of discards on the Baltic ecosystem and help to elaborate appropriate regulations on this type of dumping.

A compilation and mapping of the areas subjected to permanent and/or temporary oxygen depletion in the entire Baltic Sea area is needed. Dumping of fish remnants in these areas may be a subject of regulatory measures, if this is shown to be necessary.

The ACME also encourages the Working Group on Seabird Ecology (WGSE) to continue work related to the consumption of fish offal and discards by seabirds in the Baltic Sea.

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12 SEABIRD ISSUES

12.1 Effects of Contaminants on Seabirds

Request

This is part of continuing ICES work to improve the tools available for monitoring and interpreting the biological effects of contaminants in the marine environment.

Source of the information presented

The 2000 report of the Working Group on Biological Effects of Contaminants (WGBEC) and ACME deliberations.

Status/background information

Organic contaminants

In its 1999 report, the ACME included a review by the Working Group on Seabird Ecology (WGSE) on seabirds as monitors of marine contamination (ICES, 2000). However, the monitoring of biological effects of contaminants was only considered briefly. In 2000, WGBEC reviewed progress in studies of biological effects of contaminants in seabirds. After considering this material, the ACME agreed to the following summary of this review.

Seabirds accumulate polyhalogenated aromatic hydrocarbons (PHAHs) to a relatively great extent and these compounds can affect reproductive outcomes in heavily polluted areas. Early embryonic life stages in particular are sensitive to effects of PHAHs and bioassays should include these stages. Furthermore, there are large differences (up to a factor of 500) between species with regard to their sensitivity towards PHAHs and these differences have to be accounted for when measuring effects in standardized bioassays. A number of potential biomarkers for measuring effects of contaminants in the field have been investigated. EROD induction is one of the most sensitive markers for effects of PHAHs in birds and relationships between EROD and the reproductive success of a colony have been established. As an alternative to EROD measurements in liver tissue, a non-destructive method is also available. This method is based on the analysis of PCB patterns in blood that reflect the activity of certain cytochrome P450 enzymes including those that are linked to the Ah-receptor pathway and are associated with toxic effects of PHAHs. The method can be applied when dealing with rare or endangered species or for the analysis of time trends in individuals. Another promising tool is the measurement of carboxylated porphyrins in bird blood, as a biomarker of PCB exposure (Kennedy *et al*., 1998).

Regarding the effect of endocrine disruptors, the endocrine system of birds is well understood. As such, birds are good model organisms to study effects of contaminants on the endocrine system.

Metals and organometals

Two contaminants of particular concern in seabirds are the heavy metals mercury (Hg) and cadmium (Cd) (Furness, 1993). This is mainly because they are extremely toxic, are known to cause harmful effects to humans, and also constitute a potential hazard for marine species. They have no known biological function and anthropogenic inputs are generally more important than natural sources in their biogeochemical cycle. The uptake and bioaccumulation of these metals by seabirds are thus of interest from two points of view. First, because as long-lived species, which are at the apex of the food chain, they accumulate high levels of both Cd and Hg (Honda and Tatsukawa, 1989; Muirhead and Furness, 1988; Walsh, 1990; Stewart *et al*., 1999) and thus reflect slight variations in environmental metal levels. Second, because these high levels raise the question of the possible harmful effects of these contaminants on seabird populations. Evidence that Cd and Hg at concentrations found in pelagic seabirds cause damage to the kidney at the cellular level has been published by Nicholson and Osborn (1983).

Nevertheless, seabirds suffer from several apparent drawbacks of current biomonitoring practices (Furness, 1993; Monteiro and Furness, 1995; ICES, 2000). Sampling can be difficult and most of the time gives rise to ethical problems; thus, most studies have been conducted with seabirds that have died naturally rather than with fresh material. The use of feathers as a non-destructive technique to measure heavy metal exposure is an alternative method, especially for mercury. Furness *et al*. (1986) suggested that body feathers provide the most representative sample for estimating whole-bird mercury content. This technique (which implies knowledge of the moult cycling of the species) has allowed estimation of long-term Hg trends (Monteiro and Furness, 1997; Thompson *et al*., 1993, 1998) as well as geographical variations (Furness *et al*., 1995; Wenzel and Adelung, 1996).

The non-destructive determination of Cd is a particular problem because this metal is not transferred to any appreciable extent to feathers. Age, which is known to be a very important factor in the bioaccumulation of trace elements in other marine vertebrates, can only be determined in adult seabirds by way of a durable identification marker. Therefore, investigations into age-related changes in heavy metal concentrations are scarce, but no evidence of increasing Cd concentrations with adult age in liver or kidney has been found (Stewart and Furness, 1998) and age-class variations in both Hg and Cd levels are contradictory.

Diet is probably one of the main factors influencing bioaccumulation in seabirds and numerous studies have shown its influence (Stewart *et al*., 1997, 1999; Monteiro *et al*., 1998). As an example, Stewart *et al.* (1999) have shown that species of Procellariiforms (e.g., albatrosses, shearwaters, and petrels) from New Zealand that included appreciable amounts of crustaceans in their diet had lower Cd concentrations in liver tissue, and lower Hg concentrations in both liver and kidney tissue, than those that tended to rely predominantly on squid and fish alone. This implies that when using seabirds as monitors of trace element contamination, their diet should be well known. In this context, the choice of species will be important: Procellariiforms are suitable to discriminate environmental variation in trace element levels of oceanic food chains, whereas the larids (e.g., terns and gulls) and the alcids (e.g., auks, puffins, and guillemots) are better for monitoring trace elements in coastal environments (Monteiro and Furness, 1995).

Due to the difficulties of sampling live seabirds, little is known about the toxic effects of metals in this group. Cadmium has been shown to cause kidney pathology, and mercury can affect reproduction in some species. There have also been measurements of metallothionein (MT) induction in some seabirds, although it is not known whether these reflected harmful concentrations of metals. There may be some scope for measuring MT in seabird blood, and it may be sensible to measure selenium simultaneously as it is involved in metal detoxification through sequestration in the liver. As with PCBs (Kennedy *et al*., 1998), some effects of Hg and Cd (interference with haemoglobin synthesis) can be revealed by measuring proto-porphyrins in blood. However, seabirds are good at tolerating mercury and cadmium, as they can efficiently detoxify and excrete them, and it seems likely that Hg and Cd concentrations are generally below danger levels in seabirds. Furthermore, nothing is known about the effects in seabirds of organometals other than organomercury, although organotin compounds are also known to bioaccumulate in some species (Guruge *et al*., 1996, 1997; Kannan and Falandysz, 1997). It is therefore clear that much more research is required on metal effects in seabirds.

Monitoring biological effects of contaminants in seabirds

At present, a number of national and international monitoring programmes in the ICES area include measurements in seabirds, but they focus almost entirely on contaminant residues rather than effects. However, it should be noted that several programmes monitor such issues as reproductive success in certain seabird populations without attempting to make any link with contaminants.

With respect to whether monitoring biological effects in seabirds can add to the understanding of effects of contaminants in the marine environment, and to what the advantages of seabird monitoring may be, the ACME accepted the WGBEC conclusions that:

- some seabirds fulfill the basic requirements for monitoring species;
- they have unique metabolic features, and effects in seabirds may therefore differ from effects in fish, molluscs, or other monitoring species in use at present;
- seabirds are looked upon as an ecologically important and highly visible group of considerable concern to the public;
- seabird monitoring delivers an integrated measure of effects of contaminants over a relatively large area (approximately 1 km^2 to 100 km^2), depending on the feeding habits of the species of concern;
- seabird monitoring can include reproductive effect measurements at the individual level and as such increases the discriminatory power of effect studies;
- networks of amateur seabird enthusiasts can provide assistance with some carefully planned field studies;
- seabirds act as sentinel species which represent top predators including humans, but are more easily sampled than other top predators such as marine mammals.

Arguments against routine monitoring of biological effects in seabirds include the following:

- seabirds may travel over large distances and as such are not necessarily representative of the area where they are sampled;
- destructive sampling is not possible for rare and endangered species (and is problematic for species attractive to the public).

Need for further research or additional data

The possible monitoring of biological effects of contaminants in seabirds in national and international marine monitoring programmes should be investigated further. The following suggestions are made for research which could assist in making decisions about their more widespread use in marine monitoring:

- improve non-destructive or minimally destructive sampling involving blood, uropygial oil, skin, feathers, and eggs;
- establish linkage between abnormal behaviour and contaminants;
- develop more biomarkers in seabirds (e.g., vitellogenin);

- develop bioassays with greater environmental relevance for seabirds;
- define specific sensitivity of different developmental stages in seabirds;
- identify suitable monitoring species based on migration habits, feeding preferences, action radius, etc.

Recommendations

ICES recommends that efforts be made to set up pilot schemes for monitoring the biological effects of contaminants in seabirds, in order to obtain sufficient data for evaluating the scientific basis for using seabirds as a tool in national and international monitoring programmes. For convenience, these pilot schemes could build upon existing programmes which monitor contaminants in seabirds (eggs), or on those programmes which monitor the reproductive success and other biological features of certain seabird populations.

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12.2 Effects of Fisheries on Seabird Communities

Request

The examination of the role and interactions of seabirds in marine ecosystems is of interest to ICES Member Countries as well as to regulatory commissions.

Source of the information presented

The 2000 report of the Working Group on Seabird Ecology (WGSE) and ACME deliberations.

Status/background information

The ACME noted the growing interest with regard to the role of seabirds in marine ecosystems, and the interactions between fisheries and seabirds. At its 2000 meeting, WGSE undertook work on the effects of fisheries on the composition of seabird communities. The ACME reviewed this material, which is summarized below.

Introduction

There are direct and indirect effects of fisheries on seabirds. Most direct effects involve killing of seabirds by fishing gear or by culling, while indirect effects mostly work through alterations in the food supplies of birds (Tasker *et al.*, 2000). Seabird mortality in longline fisheries or in other fishing gear can lead to drastic population declines and may bring certain vulnerable species to the brink of extinction, but even the persistent disturbance of birds due to some fishing activities, such as some aquacultural pursuits, may negatively affect seabird numbers (Davidson and Rothwell, 1993). On the other hand, seabirds may also benefit from fisheries, because many fishing activities increase the food supply or enhance the availability of prey for seabirds. The practice of discarding unwanted fractions of a commercial catch is clearly beneficial for scavenging seabirds (e.g., Camphuysen *et al.*, 1995). Furthermore, major shifts in fish stock composition, for example due to overfishing of large predatory fish, have led to a relative increase in smaller fish, suitable for consumption by seabirds. The most prominent fishery effects have recently been summarized by Tasker *et al.* (2000).

Proving the scale of fisheries effects can be difficult due to confounding and interacting combinations with other anthropogenic effects such as pollution, culling, hunting, disturbance or with more natural oceanographic factors that can influence prey availability. In fact, fishery effects can be masked completely in seabird populations that are subject to major shifts due to these and other factors. Moreover, the life history patterns of seabirds can also buffer them to some extent from anthropogenic influences associated with fisheries.

Seabird communities

There are two main types of seabird communities. Seabirds may compete for or share breeding sites (*nesting community*) because of overlapping nesting habitat requirements or because they share a feeding area. Seabirds that share the same nesting area may, however, forage in widely separated areas and have a completely different prey spectrum, and some seabirds that nest far apart may interact while foraging in overlapping areas and perhaps even directly compete for prey (*feeding community*). While competition for nesting sites and for prey are perhaps the most thoroughly studied aspects of interspecific interactions between seabirds, the activities of one species may also enhance the feeding opportunities of another. As a result, a shift in a seabird community through a change in the abundance of one species (for example, influenced by fisheries) may affect another species simultaneously, even though that fishery plays no obvious role in the feeding ecology of the latter. For example, social-feeding, pursuit-diving common guillemots and razor-bills significantly enhance the feeding opportunities of surface-plunging kittiwakes (Camphuysen and Webb, 2000). Hence, a mass mortality of auks in gillnets could indirectly have a negative impact on kittiwakes even though they themselves do not get entangled. Also, seabirds sharing the same breeding community (e.g., terns and gulls or gannets and common guillemots), but differing in feeding ecology, may compete for nesting space, so that the foraging success of the one species may indirectly affect the nesting space of the other (e.g., Howes and Montevecchi, 1993).

So, while species in a given area may share either a certain nesting community or a feeding community or both, any aspects affecting a given seabird species may work through community interactions to affect other species indirectly. As the effects of fisheries on seabirds are usually ambiguous, WGSE concentrated on the examination of clear trends in population levels, downward or upward, on a moderately large spatial scale (e.g., southeastern North Sea, Shetland area, British Isles, etc.) for groups of species that share particular prey and foraging techniques. Such changes should lead also to shifts in relative abundance and species composition within seabird communities.

The difficulty of detecting fishery effects

From the above it is clear that fisheries effects may act directly on a species or indirectly through the wax or wane of either a competitor or a "cooperator" in mixed feeding systems. All other factors being equal, the numbers of seabirds breeding or feeding in a given area should reflect the carrying capacity of that region in terms of the amount of food available. Prey availability is not the same as the size of prey stocks present, for several factors influence the accessibility of prey for seabirds and the profitability (in terms of intake rates achievable) of a given area. Prey availability may fluctuate independently of prey stocks and prey availability is also different for species using different foraging and feeding techniques.

Fishery effects can be strong, so that populations grow or decline, but fishery effects are often indirect and may be subtle, for instance the reproductive output, activity patterns or time budgets of birds may alter. Fishery-induced increases in food supply often result in an increase in "secondary prey", that is not preferred when "normal" prey is sufficiently abundant (e.g., Furness and Hislop, 1981). A clear-cut negative effect of fisheries, such as the by-catch of large numbers of seaducks, may be very hard to quantify, because a complete census of birds that "disappear" to breed in vast Russian and Scandinavian forests is simply not feasible. Perhaps most important, however, is that fishery-induced changes in fish stocks are often very difficult to distinguish from natural variation or environmental impacts on fish, so that the next step, an effect on fish predators (such as birds), will become even more obscure.

Conclusion

Both negative and positive impacts can occur at multiple spatial and temporal scales. Commonly, effects on population abundance are difficult to demonstrate, even if they are very likely to exist. To make things more complicated, a single fishery can potentially have simultaneous positive and negative impacts on a species of seabird. For the North Sea, for example, the potential effects of shifts in age- and size-structure of fish stocks as a result of overfishing large predatory fish are very difficult to quantify, but are likely to be or have been substantial. Discards and offal as an extra source of food probably have mostly significantly positive effects for birds like the northern fulmar and several species of gulls.

Need for further research or additional data

As it is very difficult to show cause-and-effect relationships between fisheries and seabird community composition change, when there are significant fisheries closures, the opportunity should be used to study the effects of such closures on other marine communities such as seabirds. Furthermore, more background monitoring of seabird populations should be encouraged so as to be better able to follow trends.

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13 ISSUES REGARDING MARINE MAMMALS

13.1 Triennial Review of Populations of Marine Mammals in the Baltic Sea

Request

Item 1 of the 2000 requests from the Helsinki Commission: to evaluate every third year the populations of seals and harbour porpoise in the Baltic Sea, including the size of the populations, distribution, migration, reproductive capacity, effects of contaminants and health status, and additional mortality owing to interactions with commercial fisheries (by-catch, intentional killing).

Source of the information presented

The 2000 report of the joint meeting of the Working Group on Marine Mammal Habitats (WGMMHA) and the Working Group on Marine Mammal Population Dynamics and Trophic Interactions (WGMMPD) and ACME deliberations.

Status/background information

13.1.1 Grey seals (*Halichoerus grypus*)

13.1.1.1 Population discreteness, distribution, and migration

Grey seals probably entered the Baltic Sea area around 10 000 yr BP (Before Present), and they dominated central and southern areas until about 2000 yr BP. Hunting has had a major impact on this population. Numbers were substantially reduced in the Kattegat area by approximately 1850. The introduction of modern rifles and national bounty programmes in the beginning of the twentieth century resulted in more significant population declines, which were somewhat mitigated by general ice conditions that affected the availability of the herd to hunters. However, hunting was intensified in the 1930s when a series of warm winters concentrated grey seals in limited ice areas of the Bothnian Bay.

In Poland, as estimated from hunting statistics, there were approximately 1000 grey seals in 1881. The population disappeared between 1930 and 1950, and hunting was considered to be the main reason. Also along the German Baltic coast, grey seals disappeared around 1930. In both areas, grey seal sightings have increased in recent decades, but it is difficult to account for potential increases in sighting effort.

Extensive Estonian tagging programmes in 1990–1993 (number of tagged seals = 1073) provided information on grey seal movements between regions. Approximately 10 % of the tags have been recovered, primarily from the eastern central and southern parts of the Baltic Sea. About 80 % of the recoveries were from incidentally caught animals in fyke nets (eastern central Baltic) and salmon drift nets (southern Baltic). About 20 % of the returns were from observations which included live animals. Most recoveries were from pups, none from six-year old or older seals. Tagging was also conducted in Finland (number of tagged seals = 900), but it stopped in 1993. Three tags (two from Estonia, one from Finland) were recovered in the Kattegat area off the Danish coast.

Recently, breeding of grey seals has been discovered on some Estonian islands. The fast ice/pack ice interface is the preferred grey seal ice-breeding habitat. An analysis of ice cover data collected since the 1700s, indicated that the ice edge usually formed around the central Baltic, with the eastern edge not far from the Estonian islands. Animals might have aggregated here in winter. The proximity of the Estonian islands to this area might account for recent observations of breeding on these islands. No historical records are available of breeding on these islands.

Until their extirpation in Poland and Germany, grey seals also used the ice edge in these areas for breeding, especially in the shallow bays, which would provide ice even in less severe winters. In winters without ice, breeding possibly occurred on land.

13.1.1.2 Effects of contaminants, reproductive capacity, and health status

PCBs are decreasing more slowly in grey seals than in other Baltic Sea biota. However, a temporal trend analysis for grey seals in the decades 1977–1986 and 1987–1996, and a similar analysis for animals born before 1980 and animals born in 1980 and later, revealed a positive trend in gynaecological health of grey seals during these two decades, with a decrease in prevalence of uterine obstructions from 42 % to 11 %, and an increase in pregnancies from 9 % to 60 % (Bergman, 1999). The high incidence of uterine tumours (leiomyomas) seems to have decreased slowly from 53 % to 43 %. Further, Bergman (1999) also pointed at an increased prevalence of colonic ulcers in young animals, and he indicated that this might be caused by new or increased amounts of unidentified toxic factors in the seal's food. Although some of the classical contaminants were declining, some of the newer contaminants, such as the organobromines, should be examined.

In grey seals, the negative effects of contaminants on reproductive rates appear to have been reversible. The question of uterine disorders and their impact on population reproduction in this species needs to be re-examined (Bergman, 1999). Sample sizes examined as yet are small, but this may be compensated by pooling samples from the various countries for a thorough re-examination of population reproductive capacity.

Experimental work has shown cause-effect relationships between contaminant levels and skull lesions mediated through hormone secretions by the adrenal gland (Lohman *et al*., 1998).

13.1.1.3 Survey methodology and current abundance

Information on total counts of Baltic grey seals was presented to the International Conference on Baltic Seals in Pärnu, Estonia in 1999 (Anon., 1999). Total counts were 5300 specimens in 1994 and 7600 in 1999. However, these figures cannot be used to calculate exact rates of increase, because the methods have changed between years.

Currently, boat and aerial counts are used in Sweden, ground counts in Estonia, and mostly aerial counts in Finland. Pooling of the counts is complicated by the differences in methodology between areas. All regions appear to have used maximum counts before 1999 when arriving at total counts. This is because the maximum counts tended to be more precise, but they also may have led to double counting as animals move between colonies. The possibility for double counting is a major concern for the survey methods presently applied. The areas where double counting may occur is the region of Estonia – southwest Finland – Åland – southeast Sweden, where roughly 30 % of the population is found. Other areas include the North Quark between Finland and Sweden, and the eastern Gulf of Finland between Finland and Russia. Surveys should therefore be coordinated and synchronized in all countries.

Because of complications associated with double counts, and no correction for animals in the water, the moult counts are less useful for providing an estimate of absolute population size. However, they probably describe population trends fairly well. In Sweden, considerable work has been undertaken to “ground-truth” aerial and ground counts. Similar effort has started in other areas to cross-check counts from aerial, shore-based, and boat observations.

A photo-identification project was started in 1994 in Sweden. A preliminary analysis provided an abundance estimate for the three major Swedish haul-out areas that is almost two times higher than the moulting counts from those areas. Although some additional refinements are still needed in the model used, this technique is a major advance in estimating absolute population size, and this effort should be extended to encompass the whole grey seal range in the Baltic.

13.1.1.4 By-catches and other human-induced mortality

Since the 1970s, hunting pressure has been reduced, and incidental catches form an important component of human-induced mortality. In Sweden, interviews with fishermen revealed a minimum of 176 seals incidentally caught in 1996. An extrapolation to the whole fishery indicated that at least 400 seals were incidentally killed annually. The majority of these by-catches occurred in salmon gear in the Gulf of Bothnia.

In Finland, mail surveys and interviews with fishermen were conducted to evaluate seal damage to fish from July 1997 to October 1998. Survey results indicated that 37 grey seals drowned in fishing gear, and 73 % of these were in the Gulf of Bothnia.

By-catch in Poland occurs at low levels and is dominated by young juveniles. Most of the by-catch in Polish waters occurs from April to June in salmon semi-drift nets and bottom gillnets.

In Latvia, by-catches have been increasing in recent years. About 90 % of the by-caught seals occur in fish traps. The other 10 % are mostly juveniles and are caught in eelpout fyke nets and anchored salmon gillnets. Most of the by-catch occurs in spring, mainly from April to May.

In Estonia, interactions between grey seals and fishing gear increased near the end of the 1980s. This may have been related to changes in fish stocks, changes in fish movements, or to increases in the number of seals.

In 1999, 120 hunting licenses were issued in Finland (60 on the mainland, 60 in Åland). Catch data and biological samples will be made available.

13.1.1.5 Population status

The total number of grey seals counted on the Swedish Baltic coastline was 2700–3400 in 1997–1998, based on 2100–2500 for the four largest colonies north of 59 °N latitude and 600–900 for the six largest colonies south of 59 °N. The average growth rate based on the ten colonies was 6.6 %.

During the 1990s, a rapid increase in the grey seal numbers in parts of Finland has been recorded. From 1991–1999, total numbers along the southwestern coast increased from 400 to 2200. This represents an annual average increase of 24 %. Annual growth rates of up to 50 % are reported from parts of Estonian waters. Modelling and empirical data show that maximum long-term growth rates in grey seals cannot exceed 11 % per year (Harwood, 1978). These reported growth rates from the Baltic, therefore, demonstrate the need for coordinated surveys throughout the Baltic area.

By-catches are reported from all Baltic areas where grey seals and net fisheries co-exist, and by-catch levels appear to be increasing, particularly in coastal waters of the southeastern Baltic, where fisheries are being developed or expanded. Only from Sweden was an estimate of the total by-catch of seals in a fishery made available.

Reconstruction of the historical population size was carried out to provide minimum original population estimates of around 100 000 in the year 1900. Reproductive changes were taken into account in the back calculations. Recent surveys indicate that the population is recovering, but at levels approximately one-tenth of the population size 100 years ago. Rates of recovery and absolute abundance are not available due to methodological inadequacies. Analyses of the impact of contaminants on reproductive capacity, and thus on population recovery rate, are hampered by the small sample size for thorough re-examination. The population level impact of by-catches cannot be evaluated unless estimates of total by-catches are made available.

13.1.2 Harbour seals (*Phoca vitulina*)

13.1.2.1 Population discreteness, distribution, and migration

Harbour seals colonized the Kattegat and the western Baltic Sea beginning approximately 6000–8000 yr BP. Seals subsequently expanded to the east into the Kalmarsund and Gotland area. The Kattegat population was then extirpated around 5000 yr BP, leaving only eastern Baltic Sea animals (the ancestors of the present Kalmarsund population). The Kattegat area (and subsequently the West Baltic) was again colonized around 250 yr BP by animals from the Norwegian coast. As a result, Kalmarsund harbour seals differ genetically from the current Kattegat and West Baltic harbour seal populations (Stanley *et al.,* 1996; Goodman, 1998).

Beginning around 1850, the Kalmarsund population increased. There could have been 5000 harbour seals in the population around 1900. The population declined to around 1000 animals during the next decade due to hunting. Numbers remained at this level through the 1930s, but declined to the low hundreds through the mid-1970s.

Sightings of harbour seals in Poland are rare. In Germany, thirty-five sightings of seals alive, and 22 dead seals were reported during the 1990s.

Presently, there are three groups of harbour seals in the Baltic Sea environs. These include 1) Kalmarsund, 2) West Baltic, and 3) Kattegat/Skagerrak/Danish Straits. The Kalmarsund population is genetically different from other harbour seal populations, and is a discrete genuine Baltic population. No evidence exists that harbour seals have ever inhabited the Baltic Sea north of Gotland.

13.1.2.2 Effects of contaminants, reproductive capacity, and health status

Harbour seals may have been less impacted by contaminants compared to the other seal species in the Baltic Sea. However, evidence was found of increased exostosis manifested as a problem with mineralization of the bones (e.g., the lower mandible; Mortensen *et al*., 1992). By the late 1980s, the incidence had risen to 50 % of the animals examined in the Skagerrak. Levels there tend to be lower than those to the east in the Kattegat and the Baltic Sea. Similar temporal and spatial clines were found in another type of bone lesion (paradontitis).

The evidence of reproductive disorders due to contaminants is sparse for harbour seals in the Baltic Sea. However, very few animals have been examined. Although it has been shown experimentally that harbour seals fed contaminated herring had severely depleted reproductive function (Reijnders, 1986; De Swart *et al.,* 1994), there is no direct evidence for reduced capacity in the Kalmarsund population, where ratios of pups to adult females appear to have remained constant. Trends for the Kalmarsund population during 1977–1998 for one-year old and older seals (+9.5 % per year) and pups (+11.5 % per year) were not significantly different.

13.1.2.3 Survey methodology and current abundance

In 1998, the Kalmarsund population included 270 one-year old and older seals and 55 pups, and it is increasing at around 9.5 % per year. This population does not appear to have been impacted by the 1988 phocine distemper epizootic.

The West Baltic Sea population experienced 50–60 % mortality during the epizootic. By 1998, there was a slight positive trend (+4.8 %) and the numbers totalled 315 one-year old and older seals. This trend is different from that of the adjacent Kattegat/Skagerrak/Danish Straits harbour seal populations, which are increasing at near-theoretical limits. West Baltic Sea pup numbers may be decreasing, with mortality caused by grey seal or fox predation.

The Kattegat/Skagerrak/Danish Straits population suffered severe mortality during the 1988 epizootic. The first post-epizootic surveys found around 3000 animals (range from 2715 to 3015). By 1998, the numbers had increased to around 9000 animals (range from 7900 to 9747 seals). Growth during this period was 13 % per year for the area.

Counts of one-year old and older seals in these areas were obtained using replicate (three to six) aerial surveys during late August. All sites were photographed obliquely for later counting of seals from slides. All sites were surveyed in a single day. Pups were counted from the ground at the main breeding sites during June and July using spotting scopes.

Population estimates and trends were analysed using two models:

1) continuous exponential model, unstructured;
2) discrete Leslie matrix model, stage (age and sex) classified.

These models were then used to explore the consequences of changes in demographic parameters. The 1988 epizootic provided a test in that virtually all pups of the year and adult males died, resulting in a population dominated by adult females.

A problem with assessing the population from survey data results from differential rates in hauling out. Analysis of haul-out behaviour of freeze-branded animals suggests that pups of the year haul out much less frequently than adult females, while a larger portion of the adult male population hauls out than of the adult females. Thus, the age and sex composition on rookeries prior to the epizootic was different than that afterwards. Consequently, compared with a model using the stable age structure before the 1988 seal epizootic, subsequent annual surveys overestimated population size (approximately 20 %) just after the epizootic in August 1988, and underestimated population levels for the following years (approximately 15 %) (Härkönen *et al*., 1999).

The unstructured model can be a very powerful tool and, under certain conditions, may be equivalent to the age- and sex-structured model. However, depending on the initial condition, the unstructured model may overstate the population size compared to the Leslie model. Using a range of reasonable values for survival and natality, it appears that growth rates of 13 % per year are the maximum that can be achieved in a closed population with a stable age distribution. Reported values of harbour seal population growth greater than this are suggestive of either immigration or of a disturbed population structure. For example, a population dominated by adult females with few adult males or juveniles could achieve growth rates greater than 13 % on a temporary basis.

13.1.2.4 By-catches and other human-induced mortality

Some by-catch occurs with both the Kalmarsund and West Baltic populations. In southeastern Sweden, up to twenty seals from the Kalmarsund population (pups of the year) were taken in the eel fyke-net fishery. This fishery no longer exists. However, seals are reported taken in bottom gillnet fisheries in the area. A few seals were observed by-caught in the bottom gillnet fishery in Swedish waters.

In Denmark, fishermen may be allowed to shoot harbour seals interfering with the eel fyke-net fishery. In 1999, two fishermen were each licensed to shoot up to five animals. However, it is presently unknown how many seals were actually taken.

Four harbour seals were reported by-caught in German waters during the 1990s. One was caught in a fyke net, one in a trawl net, and two in unknown gear. By-catch has occurred in Poland, but at very low levels. One seal was reported taken in a cod gillnet in 1995.

13.1.2.5 Population status

The Kalmarsund population is genetically distinct, and is a genuine Baltic population. In 1998, this population included 270 one-year old and older seals and 55 pups. Observed by-catch is currently low, and the population is increasing at around 9.5 % per year.

The West Baltic Sea population suffered 50–60 % reduction during the 1988 epizootic. In 1998, the population numbered 315 seals (excluding pups), and showed a slight positive trend (+4.8 %). This trend was less than that observed in the adjoining populations. A few animals are by-caught each year in Sweden, Denmark, Germany, and Poland. Impacts of this by-catch and the Danish licensing of takes could further reduce the population’s growth rate, and should be monitored.

The Kattegat/Skagerrak/Danish Straits population suffered severe mortality during the epizootic, and the rest-population was estimated at 3000 seals in 1988. By 1998, the population had increased to around 9000 animals. The growth rate of 13 % per year is close to the theoretical maximum growth. By-catch levels and other human-induced mortality appear to be low.

13.1.3 Ringed seals (*Phoca hispida botnica*)

13.1.3.1 Population discreteness, distribution, and migration

Ringed seals entered the Baltic around 11 000 yr BP, and are at present separated into three geographically distinct concentrations (Härkönen *et al.*, 1998), termed groups in this report: the Bothnian Bay group, the Gulf of Finland group, and the Gulf of Riga group. A genetic screening showed no significant differences among the main groups. This is not unexpected, since there was a single population covering the Baltic Sea, which only recently separated into the three groups observed today.

Time series of the population, which were reconstructed from catch data and trends, seem identical to that of grey seals. The numbers were estimated to be in the order of 200 000 at the beginning of the twentieth century, fell rapidly from 1910 to 1940, and have been at a low level since the early 1970s. The similarity in the patterns may be due to the advent of high-power rifles, and socio-economic and weather conditions, which affected both species.

In Estonian coastal waters, ten mature ringed seals were tagged with satellite-linked transmitters, and their movements (2305 locations) were compared to the movements of five satellite-tagged seals in the Bothnian Bay (645 locations). Each group exhibited limited movements within their respective tagging areas. These areas were centred in regions of ice cover, which is necessary for whelping. In this context, it is worth noting that the recent reduction of ice cover (Seinä and Palosuo, 1996) may affect carrying capacity.

Seasonal changes in diving behaviour were demonstrated in the ten seals satellite tagged in Estonian waters. About 200 000 dives were recorded, and the seals spent approximately 70 % of the time diving in July, compared to 30 % in January to March. The locations and depth of dives corresponded with the seasonal distribution of Baltic herring, which reside in shallow coastal areas in spring and in deeper offshore waters in summer.

13.1.3.2 Effects of contaminants, reproductive capacity, and health status

Toxic effects on the immune system and reproduction caused by contaminant exposure are reversible until a certain threshold has been reached (Reijnders, 1986; De Swart *et al*., 1994). PCB and DDT levels are still high in Baltic ringed and grey seals. Ringed seals suffer from a clearly higher toxic burden than grey seals. Cytochrome P4501A (CYP1A) induction is a biomarker for organochlorine exposure. CYP1A activity is elevated in both Baltic seal species compared to the Arctic ringed seals and to Atlantic grey seals from Sable Island, Canada. A gender difference in the enzyme activity is seen only in the Canadian seals, with males showing higher CYP1A activities. This could be due to two of the males having a PCB and DDT burden similar to that of the Baltic seals, and therefore also induced CYP1A activities. As females transfer a major part of their contaminant burden to their pups during lactation, the contaminant load does not increase with age, as it does in males. In the Baltic Sea, although females had lower PCB and DDT levels, their CYP1A activity was on the same level as that of the males.

Another commonly used biomarker for DDT and PCB exposure is an induced CYP2B activity. However, this enzyme does not appear to be present in ringed and grey seals (Mattson *et al*., 1998).

Hepatic and blubber vitamin A levels are depleted in both species of Baltic seals compared to reference populations in terms of contaminant exposure. The depletion could be due to dietary differences or to the toxic effects of the contaminant load on the vitamin A status. Dietary differences between the regions should be taken into account before ascribing the differences to contaminant effects. Vitamin E, an anti-oxidant, showed the opposite trend to vitamin A and was higher in the Baltic seals than the references. This could again be due to dietary differences between areas, or it could be a result of an increased demand for radical scavengers (anti-oxidants), as some contaminants cause the production of toxic oxygen radicals.

Concentrations of mercury (Hg), cadmium (Cd), lead (Pb), and selenium (Se) in ringed seal tissues from 37 animals (ages 0–32) caught in the Gulf of Bothnia and at Svalbard (in the Arctic) were compared to concentrations of these metals in tissues from 40 grey seals (ages 5–35) caught in the Gulf of Bothnia and at Sable Island, Canada. Concentrations of Hg and Se were considerably higher in Baltic ringed seals, but Cd was lower than in Svalbard ringed seals. There were no differences in Pb concentrations between regions. In the Baltic, the Hg and Se burdens in ringed seal livers were considerably lower than those in grey seal livers. The Cd and Pb levels were similar in both species in the Baltic. By comparison with effect threshold levels reported in the literature, only the Hg levels can be considered high.

A study of the pathology of ringed seals from Finnish coastal waters during 1982–1995 described several species of parasites that could impact the health of seals. Heartworms were the most common in young seals; up to 21 % of 104 young of the year in the Gulf of Finland were infested. Heartworms were only found in one specimen out of thirteen from the Gulf of Bothnian waters. Heartworms were not found in any specimen over three years of age. Every individual over two months of age was infested by lungworms (*Parafilaroides* sp.), gastrointestional nematodes (*Contracaecum osculatum*), and certain acanthocephalan worms (*Corynosoma strumosum* and *C. semerme*). These parasites may be significant health factors in individual animals.

Uterine occlusions, which affect only ringed seals in the Baltic, are well documented. These occlusions lead to sterility, but otherwise the animals are in good condition. This phenomenon was first seen in the early 1970s; it peaked in the late 1970s and has slowly decreased since then. It has been estimated retrospectively that uterine occlusions emerged in the Bothnian Bay group in the latter half of the 1960s (Helle, 1980a). The uterine occlusion originates from a disrupted pregnancy development and its occurrence is dependent on the age of the female. A relationship between the frequency of the occlusions and PCB concentrations is being investigated.

Population growth rate and pregnancy rate declined significantly from the 1960s to the late 1970s, then recovered in the 1990s. The recovery coincides with a decrease in PCB concentrations in herring. Bothnian Bay population estimates fell from 14 000 in the 1960s to about 4000 in the early 1980s, and a modest recovery has occurred since then.

13.1.3.3 Survey methodology and current abundance

The first comprehensive survey for the entire area was conducted in 1994–1996. The survey was conducted as an aerial strip survey, and the method applied is well documented (Härkönen and Heide-Jørgensen, 1990; Härkönen and Lunneryd, 1992). The estimated hauled-out Baltic population of ringed seals was 5510 ± 42 % (95 % confidence interval). Of this estimate, 3945 ± 1732 were in the Gulf of Bothnia, 1407 ± 590 were in the Gulf of Riga, and about 150 were in the Gulf of Finland (Härkönen *et al*., 1998).

For the Bothnian Bay, the first surveys were carried out in 1975 (Helle, 1980b) and a decreasing trend was seen up to 1984 (Helle, 1990). Annual counts during the period 1988–1998 showed a significant increase at 5 % per year (Härkönen *et al.*, 1998).

13.1.3.4 By-catches and other human-induced mortality

In Swedish studies, only ten to twenty or fewer ringed seals were directly reported by-caught. Most reports were from the Bothnian Bay and concerned fyke nets.

In Estonian coastal waters, not more than 10 % of the total by-catch of seals is ringed seals, which would be about twenty animals.

Total by-catch reports in Finland were about thirty seals per year in earlier years, and ten in recent years. The fraction of ringed seals in these samples has fallen in recent years to 10 %. In an enquiry to fishermen on seal damage to gear and on by-catch, 56 % of the by-catch of both grey and ringed seals was found in fyke nets, 24 % in drift nets, and 20 % in other gear. Four animals were reported by-caught from Polish waters since 1995. No data are available from the Russian area.

In Finland, 99 ringed seals were collected for autopsy from 1982–1995, mostly from the Gulf of Finland (Westerling and Stenman, 1999).

13.1.3.5 Population status

The Baltic ringed seal population numbered about 200 000 in 1900. The most recent estimate (1996) for the hauled-out Baltic population was 5510 ± 42 % (95 % confidence interval). All three groups of ringed seals in the Baltic Sea are severely depressed. Based on the 1996 estimate, the Bothnian Bay group was 3945 ± 1732 and increasing at approximately 5 % per year. The Gulf of Riga group was 1407 ± 590, and no information on trend is available. For the Gulf of Finland, the 1996 hauled-out population was about 150 seals; the 1999 estimate was 150–300, however, no significant trend was documented.

Population status is affected by ice conditions and contaminants and environmental conditions are felt to be more important than direct human-induced sources of mortality for the rate of recovery in this population. However, reports indicate that by-catches are widespread, but no estimate of total by-catches based on observer data is available for any of the three ringed seal groups.

13.1.4 Harbour porpoises (*Phocoena phocoena*)

13.1.4.1 Population discreteness, distribution, and migration

A new method to distinguish between putative populations of harbour porpoises has been introduced (Lockyer, 1999). This method uses tooth ultra-structure, and is based on the fact that teeth in harbour porpoises continue growing throughout life and therefore can provide a permanent record of life history events. Nine different characters were identified as potentially useful in the decalcified, sectioned and stained teeth, and scores for each character were compared using Chi-squared analyses for a variety of geographical regions throughout the North Atlantic and also from California in the Pacific. Specific investigation of teeth samples from the ASCOBANS region indicated differences between areas within the North Sea, and between the North Sea; the Skagerrak, Kattegat, and Inner Danish waters; and the Baltic Sea.

Morphometric skull characteristics showed significant differences between porpoises of the central Baltic Sea (Arkona Sea and waters off eastern Sweden); the Skagerrak, Kattegat, and Inner Danish waters; and the German Bight (North Sea). Statistical analysis (ANOVA, Discriminant Analysis and Chi^2-tests) compared cranial measurements and non-metric characteristics of 242 harbour porpoises. The results of this study, together with tooth analysis, confirmed the difference between Baltic and North Sea porpoises, and further indicated differences between the animals from the transition zone (Skagerrak through the Inner Danish waters) and the central Baltic Sea, respectively, indicating the existence of a separate population in the Baltic Proper. In the cranial study, female harbour porpoises exhibited more powerful statistical results for a separation into discrete populations than males. These results did not show support for seasonal migrations, as discussed by some authors (e.g., Kinze, 1990). However, if earlier reported large-scale migrations from the Baltic to adjacent sea areas also apply to the present situation, the suggested population delimitation might be too rigid.

The levels of heavy metals and metals (Hg, Se, Cd, Pb, Cu, Zn, Cr, Ni, Mn, and Fe) in tissues of harbour porpoises from Danish waters and Polish coastal waters showed that levels were correlated with age for the specimens studied, and that concentrations were higher in Danish waters than in the Baltic Sea. In general, these results supported the findings of a discrete Baltic population.

13.1.4.2 Effects of contaminants, reproductive capacity, and health status

Mercury (Hg) is regarded as highly toxic and has been linked to immunosuppression and disease in mammals. Harbour porpoises stranded on German North Sea and Baltic Sea coasts showed decreased nutritional condition and increased pathology of the respiratory system with increased mercury levels (Siebert, 1995).

In general, reproductive parameters in harbour porpoises are well studied, but there remain several aspects that are uncertain: for example, the duration of pregnancy, weaning, and lactation. The reproductive cycle is markedly seasonal (Sørensen and Kinze, 1990, 1994), and in theory ovulation and pregnancy may be feasible yearly. However, the likelihood is that the true reproductive interval may vary from one to two years. Lockyer and Kinze (1999) estimated that even if females have a longevity of 20 years with an age at sexual maturation of 3–4 years, the maximum expected number of young produced in a lifetime might only be 11–12 calves. For most females, longevity does not exceed ten years, so that only five young might be produced.

Information on the reproductive capacity of porpoises is almost totally lacking in the Baltic, and although effort is, and has been, directed there, the low abundance of porpoises makes any study very difficult.

13.1.4.3 Survey methodology and current abundance

The most recent information on the abundance of porpoises in the Baltic was presented in ICES (1997). New information concerning abundance and distribution is essential in order to assess the impact of by-catches. At very low population densities, it is expensive and difficult to estimate cetacean abundance from conventional line transect surveys. A working group under ASCOBANS has been considering the feasibility and design for a prospective new survey in the Baltic. This group should be encouraged to complete its work.

13.1.4.4 By-catches and other human-induced mortality

Independent observers monitored the Danish bottom-set gillnet fleet in the Baltic Sea between 1992–1998. No by-catches were recorded on the vessels carrying observers (Vinther, 1999). However, by-catches were recorded within the Kattegat and the North Sea. In the North Sea area, where most effort had been deployed during the same period, most by-caught porpoises occurred in the cod and turbot fleet and were young juveniles (Lockyer and Kinze, 1999), with the highest numbers in the first and third quarters of the year (Vinther, 1999). Estimates of extrapolated total by-catches were only possible for that particular fishery in the North Sea.

Between 1990–1999, 44 porpoises were recorded by-caught off the Polish coast. Salmon semi-drift nets were responsible for 41 % (18) of the by-catches and bottom-set nets for cod were responsible for 34 % (15). Up to 14 % (6) of the by-catches were in other bottom-set nets and the remainder in trawls. It was noted that the age distribution of the by-caught animals ranged from 0–6 years, with nearly all animals being in the age group 0–2 years. The highest by-catches generally occurred during the months when the salmon fishery was operational (December–April) within the Puck Bay area.

13.1.4.5 Population status

All available information indicates that the abundance of porpoises in the Baltic Sea is reduced and at present very low. The most recent information on abundance is contained in the 1997 ACME report (ICES, 1997).

This very depleted population is subject to an unknown level of by-catches, and estimates of abundance and by-catch levels are urgently needed before the status of porpoises in the Baltic Sea can be evaluated.

Need for further research or additional data

For further research and monitoring of marine mammals in the Baltic Sea, the ACME recommends the following actions.

Grey seals

- Survey methods should be standardized and surveys should be synchronized to obtain reliable estimates of abundance and trends.
- Efforts to obtain photo-identification data should be expanded to encompass the full range of Baltic grey seals to allow for independent estimation of absolute abundance.
- There should be international coordination and pooling of samples for a re-examination of the effects of contaminants on reproductive capacity and on population recovery rate.

Harbour seals

- The surveys to monitor abundance and trends should be continued on a routine basis, using well-established methods with aerial surveys and photographic documentation of the observations.
- Monitoring of direct and indirect human-induced mortality should be established.

Ringed seals

- The surveys to monitor abundance and trends should be continued on a routine basis, using well-established methods with aerial strip surveys and photographic documentation of the observations.

- Research on effects of contaminants on the reproductive capacity and health status of Baltic ringed seals should be continued and augmented.

Harbour porpoises

- Methods for obtaining reliable abundance estimates in low density populations should be developed and surveys deploying relevant methods should be planned and implemented.
- Areas where there are high frequencies of by-catches should be identified, and possible mitigation measures should be explored.
- A regime to collect by-caught porpoises for studies of population structure and health aspects should be established.
- Coordination and collaboration should be sought with the IWC Scientific Committee's research programme to establish cause-effect relationships between contaminants and population-level effects in cetaceans.

Recommendations

ICES recognizes that the marine mammal populations in the Baltic Sea are severely depleted compared to population levels 100 years ago. ICES is aware that the severely depleted populations are suffering from unknown levels of by-catch mortality. Under the present situation, the seal populations may be recovering, at least in part of the Baltic Sea area. However, the abundance of the harbour porpoise is unknown but assumed to be at very low levels, and any by-catch may pose a threat to the viability of this species in the Baltic Sea. Therefore, ICES recommends that the appropriate management authority in the respective Member Countries establish systems for obtaining estimates of total by-catch of all marine mammals in their Baltic Sea fisheries, and that relevant by-catch mitigation measures be implemented with particular emphasis on localized areas where by-catches are assumed to be high.

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13.2 Research Programme on Cause-Effect Relationships between Contaminants and Population-Level Effects in Seals

Request

This is part of continuing ICES work to investigate cause-effect relationships between contaminants and population-level effects in marine mammals.

Source of the information presented

The 2000 report of the joint meeting of the Working Group on Marine Mammal Habitats (WGMMHA) and the Working Group on Marine Mammal Population Dynamics and Trophic Interactions (WGMMPD), the 1999 report of the Working Group on Marine Mammal Habitats (WGMMHA), and ACME deliberations.

Status/background information

In 1999, WGMMHA developed a plan for research on cause-effect relationships between contaminants and population-level effects in seals. WGMMHA identified a sub-group to complete the proposal. The ACME noted that the project documentation is now available.

The specific aims of the submitted proposal are:

- to determine the population dynamics, health status, habitat use and diet, tissue concentrations and kinetics of contaminants in harbour seals (*Phoca vitulina*) inhabiting five areas representing a gradient of contamination;
- to develop and validate a set of functional responses to contaminant exposure in captive animals and utilize them in free-living populations in a gradient of contamination;
- to combine the results obtained to develop a set of interrelated models aimed at providing the basis for advice for management of harbour seals and their habitats in European coastal marine ecosystems.

The harbour seal was chosen as the model species because it is a common top predator in large parts of coastal Europe, and it is a resident species inhabiting a significant gradient of contamination within this area. Five populations were selected for the study: the Dutch and German Wadden Sea, Skagerrak/Kattegat, Moray Firth, and central Norway. These areas were chosen because they represent a contamination gradient, and the relevant background information on demography and time series of population dynamics are available. The proposal is an innovative approach to addressing the cause-effect relationships linking environmental contaminants to population-level effects in a top predator.

The project includes thorough studies of habitat use and foraging, contaminant levels in predator and prey species at the actual foraging grounds, current health, demography, and trend analyses in five free-living populations in a gradient of contamination. The research on free-living populations will deploy non-invasive techniques. This part of the project will, *inter alia*, address the variability of contaminant exposure observed between animals within the same local population, and relate this to individual behavioural traits.

The experimental research on captive animals will develop non-invasive techniques to validate biomarkers (indicators of functional response in reproduction, immune, and endocrine systems) for application in free-living seals.

The results of the studies on free-living and captive seals will be used to establish a set of interrelated models. A holistic conceptual model will link sub-models ranging from spatial GIS models of exposure to predictive mathematical risk-assessment models that will indicate likely effects on individuals and populations at various levels of exposure.

Recommendations

The ICES ACME approves the research plans for establishing cause-effect relationships between environmental contaminants and population-level effects in marine mammals, and stresses the relevance and importance of this research for the ability of ICES to respond appropriately to requests regarding the status of marine mammal populations in contaminated areas.

The ICES ACME therefore recommends that these research plans be implemented, and urges ICES Member Countries to facilitate the funding of this research programme.

14 ISSUES RELATED TO MARICULTURE

14.1 Environmental Interactions of Mariculture, including New Research and Monitoring Programmes

Request

This is part of the continuing ICES work to keep under review environmental issues relating to mariculture.

Source of the information presented

The 2000 report of the Working Group on Environmental Interactions of Mariculture (WGEIM) and ACME deliberations.

Status/background information

Marine fish and shellfish production continues to increase in many countries, but has been affected by disease and toxic events. For example, salmon production in Scotland, eastern Canada, and Norway has been impacted by direct and indirect effects of Infectious Salmon Anaemia (ISA). In Ireland and western Canada, salmon production has been static due to regulatory problems. Production of rainbow trout in Denmark has been severely restricted by nutrient inputs and feed quotas. In Germany, marine rainbow trout production is confined to a single site. Competition for space in the coastal zone is leading to increased interest in recirculation systems in some countries, e.g., France and Germany. Despite feed quotas and a number of ISA outbreaks, Norway continues to achieve significant increases in the production of both rainbow trout and salmon.

The production of turbot, seabream, and sea bass is expanding in France and Spain. Production of shellfish (mussels, oysters, and scallops) is also expanding rapidly. There is increasing concern over the environmental impacts of intensive shellfish farming, notably benthic impact and possibly reduced oxygen levels in the water. It is likely that regulatory authorities will require monitoring programmes to be developed to control these effects.

14.1.1 Monitoring programmes in Member Countries

The ACME noted that there are now a number of monitoring programmes in ICES Member Countries which are based largely on the work initiated by WGEIM in the late 1980s.

Initially the focus of the monitoring programmes was on the water column because of concerns that the input of waste from salmon farms could result in an increase in water column nutrient concentrations and lead to an increase in the occurrence of harmful algal blooms. Subsequent monitoring demonstrated that, in general, salmon farming activities did not lead to a detectable increase in nutrient concentrations and the focus of monitoring since then has, for the most part, shifted to the impacts on the sediments.

Monitoring programmes for finfish mariculture facilities

The first comprehensive environmental monitoring programmes for finfish aquaculture activities were developed during the early 1990s in Norway (LENKA and MOM) and in Scotland (regulation and monitoring of marine cage fish farming in Scotland). These programmes are directed at marine finfish aquaculture. The Scottish programme is overseen by the Scottish Environment Protection Agency and aims at protecting the quality of the coastal waters and regulating discharges from the fish farms. The primary focus of the programme is on the quality of water and sediments in terms of organic enrichment and nutrient release. An empirical model, DEPOMOD, has been developed in Scotland to predict the extent and effects of solids deposition from finfish mariculture on the benthos. This model can be used in the environmental impact assessment (EIA) process for site selection, determination of carrying capacity, and prediction of benthic impacts. The model then becomes a monitoring tool for comparison of predicted and existing measured effects that takes into consideration changes in fish biomass. One of the main benefits of DEPOMOD, as for most empirical models, is its potential for identifying research requirements as well as its ability to incorporate new advances from these research activities. There are plans to add a hydrodynamic model to DEPOMOD as well as a sea lice treatments component. The latter will aim at predicting concentrations in sea water for soluble treatments and in sediment and biota for in-feed treatments.

The proposed revised Norwegian programme (MOM) is a management system with a simulation model and environmental quality standards, which aims at adjusting the local environmental impact of a marine fish farm to the holding capacity of the site. This programme has three main investigation points: 1) sedimentation rate; 2) chemical condition of the sediment; and 3) benthic infauna community. This programme has a geographical component with three levels of impacted zones: local, intermediate, and regional.

An annual benthic monitoring requirement regime for marine salmon farms has recently been introduced in Ireland. It has three monitoring levels defined by the tonnage produced and the mean current speeds at the fish farm. The monitoring requirements vary from direct photographic/video observations (Level 1) to redox measurements (Level 2) to macro invertebrate fauna inventories (Level 3) at a total of six sampling stations.

The results of the annual monitoring surveys are used to determine whether or not an increase in production tonnage can be permitted at the farm site.

In eastern Canada, a benthic impact monitoring programme was adopted in New Brunswick in 1995. This monitoring programme is presently a requirement of the aquaculture site license and is conducted annually. A similar programme exists in France with mandatory sampling requirements from the farm operator and sporadic checks from regional authorities. New monitoring programmes are being developed on the west coast of Canada.

A general principle of the benthic monitoring programmes now in place is that an allowable zone of impact is acceptable and that outside this zone, typically in the region of 100 m from the farm, the benthic conditions must not differ from ambient conditions. Within the impacted zone, however, anoxic and afaunal conditions are not acceptable.

While monitoring programmes, which are largely paid for by the salmon farmers, are expensive, they are considered to be necessary for regulatory purposes and to provide the public with the necessary information to show that salmon farming has only a very localized impact and does not have a significant negative impact on the environment as a whole. Monitoring has clearly shown that salmon farming, in comparison with other activities in the coastal zone, has had an insignificant impact on the coastal ecosystem.

Potential environmental impacts of shellfish farming

Shellfish farming activities, like all other activities in the coastal zone, can have an impact on the environment. The impacts can be both positive and negative and can include visual impacts, impacts and interactions with local wildlife, as well as impacts on the water column and sediment. The impacts, principally from suspended mussel cultivation, will be briefly examined and discussed below.

In some cases the source of mussel seed for on-growing is obtained by the dredging of seed beds. It is recognized that dredging of seed, as a fishing activity, can have a significant impact on the benthic environment, but considering the scale of this activity relative to overall commercial fishing, the impacts are insignificant.

In terms of water quality, it is generally well accepted that shellfish aquaculture can be of benefit to the coastal system, particularly in the minimization of eutrophication effects through the grazing of phytoplankton with consequent improvements in water clarity. The grazing activity of the shellfish can mitigate against other coastal activities that contribute to nutrient enrichment of these systems, such as agriculture and tourism. Potential depletion of phytoplankton is not considered to be a major concern considering that the economic carrying capacity of a system will be met significantly earlier than the ecological carrying capacity of that system. It is also well recognized that shellfish farms contribute to an increase in pelagic biodiversity and abundance for two main reasons. Firstly, the farm infrastructure acts as a reef system allowing the recruitment and growth of many epifaunal species which are prey for several pelagic species, including high value sport fish such as the sea bass and possibly some salmonid species. Secondly, grazing shellfish species are recognized as being the main source of benthic-pelagic coupling in the marine ecosystem.

There is little scientific evidence to show that suspended mussel cultivation or the cultivation of oysters on trestles can have a detectable negative impact on the water column. The deposition of faeces and pseudofaeces can, however, impact on the benthos in a similar way to the deposition of faeces and waste food from salmon farms, particularly in areas where the tide and wind-induced currents are weak. Recent studies on the sediments in the vicinity of mussel farms in Scotland and Ireland have shown that the impacts on the macrobenthos and sediment chemistry are observable but localized. Similar results were obtained in studies carried out in Sweden, Spain, and New Zealand in the 1980s.

Monitoring at shellfish production areas

It has been argued that shellfish farming is self-regulating in that decreasing growth rates because of over-stocking effectively limit the annual production. From a regulatory perspective, however, such a premise is not useful.

Environmental monitoring programmes similar to those described above for salmon farms currently do not exist for shellfish aquaculture, although the need to develop such programmes to evaluate the impact of shellfish aquaculture on the environment has been identified, particularly in France, Scotland, Ireland, and Canada. Recent mussel site allocations in Ireland have been made conditional upon monitoring requirements. These requirements include monthly assessment of starfish abundance, water temperature, salinity, dissolved oxygen, and chlorophyll *a*.

The ACME endorsed the conclusion of WGEIM that, scientifically, the focus of monitoring of the impacts of shellfish farming on the environment should be on the benthos. Monitoring of the water column, e.g., phytoplankton, chlorophyll, is not scientifically justifiable and the results of such monitoring programmes are not useful in determining impacts or in setting stocking densities and allowable annual production.

Research on modelling the benthic impact of suspended mussel cultivation

Research on modelling the benthic impact of mussel farming, primarily by adapting the fish farm impact model DEPOMOD, has recently been conducted in Scotland. While this work is at an early stage of development, the initial results suggest that the predicted rate and extent of deposition of waste could be linked in a general way to the observed impact on the sediment and the benthic fauna. This work has also highlighted the need for additional studies to provide the necessary data for input to the model. These include further data on the settling rate of faeces and pseudofaeces and on the temporal variation in seston levels.

Similar research is presently being conducted in Canada, with the added objective of incorporating a benthic primary production component. Research has also been conducted in France to determine the impact of biodeposits from oyster table systems in the region of Marennes-Oléron and suspension line culture in the Etang de Thau region.

One environmental concern that is commonly associated with shellfish farming is the changes that can occur in predator populations, e.g., starfish and crabs, as a result of "fall out" of crop. These effects, although they are not of significance to the overall productive capacity of the system, can have an effect on the survival of other species, namely "Species at Risk" (SAR). There are no scientific data available to support these types of interspecific impacts. It is recommended, therefore, that more research be carried out to investigate these perceived impacts and to formulate effective, realistic, and scientifically relevant monitoring aims to meet environmental objectives.

The ACME, recognizing the importance of making available to Member Countries all information on the results of ongoing monitoring programmes, and directions in research and monitoring priorities and public attitudes and concerns, noted the value of the proceedings of the Symposium on Environmental Effects of Mariculture that was held in St. Andrews, New Brunswick, in September 1999. The proceedings of two of the three sessions of this Symposium will be published in the *ICES Journal of Marine Science*.

14.1.2 Control of sea lice in salmon cultivation

The ACME provided information on sea lice treatment at salmon farms in its 1999 report (ICES, 2000) and took note of the updated information on this topic provided by WGEIM, as summarized below.

The ongoing need to control sea lice in salmon farms, both for husbandry purposes and to minimize the impacts on wild fish species, has led to a number of new initiatives. The most recent developments in the control of sea lice can be summarized as follows:

1) the availability of an increased range of treatment chemicals,
2) the definition of coordinated strategic approaches to lice control, and
3) the linking of lice numbers to a formal requirement for chemical treatment.

Treatment chemicals for sea lice

In Scotland, the organophosphorus chemical dichlorvos (in commercial formulation) was for many years the only treatment authorized for the control of sea lice in farmed salmon. It was administered as a bath treatment, and was subsequently released to the surrounding aquatic environment. Approximately five years ago, preparations based on hydrogen peroxide also became available. Within the past two years, dichlorvos has lost its authorization for use in fish farming. However, bath treatments based upon azamethiphos and cypermethrin have been authorized, together with in-feed treatments based upon teflubenzuron and emamectin benzoate. The industry therefore now theoretically has available a series of five alternative authorized treatments. The efficacy and target life stages vary between treatment chemicals and, thus, in practice the choice is more limited than may appear at first consideration.

In Norway, azamethiphos, cypermethrin, teflubenzuron, and emamectin are also used, together with deltamethrin.

In Ireland, dichlorvos is used both for market-size fish and for broodstock. However, its use has declined significantly in recent years. Other treatments used include azamethiphos, cypermethrin, and emamectin. Recently, teflubenzuron, sold under the trade name Calicide, has received full market authorization for use in Ireland.

In Canada, bay management schemes are being implemented where appropriate. The use of ivermectin is permissible as off-label treatment. It has a long withdrawal time, and therefore is only used on first-year fish. Azamethiphos (Salmosan) received temporary registration in 1995, and can be used in Canada. Cypermethrin has been used in the USA, but is not allowed in Canada.

One of the other principal therapeutic agents registered for use is avermectin. It has been used since 1999 on an experimental basis. Unfortunately, the formulation used, "Slice", has been removed from the emergency drug release listing and registration of a replacement therapeutant is not anticipated. This leaves a significant gap in the mariculture industry's access to therapeutants.

There is awareness that it is necessary to ensure that these chemicals are used safely, to minimize risks to consumers, farm operators, and the environment. In most countries, the authorization procedures for these substances include some element of assessment of the acceptability of the environmental risk.

In the UK, the authorization of these substances in medicines is supplemented by a requirement for the farm operator to obtain specific permission from the appropriate regulatory authority (in Scotland, the Scottish Environment Protection Agency (SEPA)) to discharge treatment chemicals to the sea. Permission is required on a site-by-site basis. If granted, the permit will define the chemicals which can be used, and the quantities that may be used within one or more defined time periods (e.g., an annual amount, and an amount that may be used per day). The amounts differ between sites; for example, a higher energy site with relatively effective dispersion may be able to accommodate a greater quantity of a dissolved treatment chemical than a less dispersive site without causing environmental damage. The determination of these amounts is based upon Environmental Quality Standards derived from ecotoxicological risk assessments. Mathematical modelling of each site, utilizing site-specific hydrographic data, of the dispersion of dissolved chemicals and of the dispersion and settlement of chemicals on particulate matter is used to derive safe quantities of chemicals.

Coordinated strategic approaches to lice control

By analogy with the strategic approaches taken to the control of certain pests and diseases in terrestrial agriculture, there is increasing interest in similar approaches to sea lice control. In Scotland, it has been noted that there is a period in the spring when the numbers of adult female lice are low, they produce few eggs, and the eggs are of relatively poor quality. The Scottish industry has developed a strategy in which all the farms within a defined area of the coast would simultaneously treat against sea lice at this vulnerable stage in their life history. Results from trial areas indicate that the level of control achieved can be high, and that in some cases, no further treatment is necessary for many months. Other advantages from this approach include increased growth rate of treated fish due to the general improvement in health and reduced impact from treatments.

The main environmental concern related to this approach arises from the use of treatment chemicals at all farms in an area simultaneously. There is a possibility that a short period of relatively intensive use of a treatment chemical might result in exceeding the Environmental Quality Standard (EQS) value for that chemical in the receiving waters. SEPA has therefore expressed a cautionary welcome for the initiative of the strategy, with recognition of the potential benefits in the long term. Further discussions are required to determine how the potential benefits of the strategy can be achieved without posing undue risk to the environment.

The Norwegian national action plan against sea lice also includes coordinated regional treatments during the cold periods of the year, when lice numbers are normally low and lice may be more susceptible to the chemical treatments concerned.

Formal linking of lice numbers to a requirement for treatment

There is a body of largely circumstantial evidence which appears to link declines in the health and numbers of wild salmonids (particularly sea trout) in some coastal areas with the presence of salmon farms, particularly when sea lice are abundant. In some coastal areas, the total number of lice on farmed salmon greatly exceeds that on wild salmonids. There are moves in some countries to establish a maximum acceptable lice burden on farmed salmon.

In Norway, for example, the control of sea lice is given a high priority. A national action plan has been developed, with the following objectives:

a) regional collaborative groups should plan and coordinate measures to control lice;

b) records should be kept of the occurrence of lice at farms raising fish for slaughter;

c) the occurrence of lice on wild fish should be documented;

d) measures taken to prevent or combat infections should be documented;

e) organized delousing during cold periods of the year shall be planned and implemented;

f) as a long-term objective, the harmful effects of sea lice on farmed and wild fish should be minimized.

Regulations have been issued defining the average number of lice per fish that will trigger compulsory therapeutic delousing at different times of the year. Synchronized delousing in an area is recommended. In southern Norway, delousing is mandatory when counts of mature females average 0.5 per fish from 1 December to 1 July, or more than five mobile stages of lice per fish. Delousing must cover all cages on a site.

The control of sea lice on marine salmon farms continues to have a high priority in Ireland. All salmon farm sites are inspected fifteen times per year, plus any follow-up inspections required where instructions to reduce lice levels have been issued. Farms are inspected twice per month during March, April and May, and once per month thereafter.

Treatment triggers during the spring period are set close to zero, in the range 0.3–0.5 egg-bearing females per fish, and are also informed by the numbers of mobile lice on the fish. Where numbers of mobile lice are high, treatments are triggered even in the absence of egg-bearing females. Outside of the critical spring period, a level of 2.0 egg-bearing lice acts as a trigger for treatment. This is only relaxed when fish are under harvest. Triggered treatments are underpinned by follow-up inspections and, where necessary, by sanctions. Sanctions employed include peer review under the Single Bay Management process, conditional fish movement orders, and accelerated harvests.

There are no statutory limits to lice numbers in Canada. Treatment is site specific, and typical triggers that might be used by farmers would be five pre-adult lice per fish or two gravid females per twenty fish.

Recommendation

ICES notes the importance of maintaining low lice levels on salmon farms and recommends that Member Countries consider, in the light of local conditions, the need to adopt coordinated strategies to control lice at salmon farms.

Reference

ICES. 2000. Report of the ICES Advisory Committee on the Marine Environment, 1999. ICES Cooperative Research Report, 239: 119–121.

14.2 Effects of Mariculture Activities in the Baltic Sea

Request

Item 3 of the 2000 requests from the Helsinki Commission: to prepare Chapter 9 on "Marine fish migratory and freshwater species in the Baltic Sea Area" for the Fourth Periodic Assessment of the State of the Baltic Marine Environment, 1994–1998.

Source of the information presented

The 2000 report of the Working Group on Environmental Interactions of Mariculture (WGEIM) and ACME deliberations.

Status/background information

In 1999, the Working Group on Environmental Interactions of Mariculture (WGEIM) drafted an outline for the preparation of material on mariculture activities in the Baltic Sea for this chapter. Although not all of the anticipated material was available in 2000, WGEIM reviewed the topic under the following issues:

- aquaculture production in the Baltic Sea;
- use of chemicals and medicines in Baltic Sea aquaculture;
- examination of the role of fish culture in eutrophication of the Baltic Sea.

Aquaculture production

Information on the production of fish and shellfish in the Baltic Sea area was obtained from the FAO FISHSTAT 1997 database. The main species produced in marine waters is rainbow trout, mainly in Finland, Denmark, and Sweden (see Table 14.2.1). A small rainbow trout industry (150 tonnes) existed in Estonia up to 1995. The only other species produced in salt water is the European whitefish, 33 tonnes of which were produced in 1994 in Finland. There is no marine production of fish or shellfish in the southern Baltic countries of Poland, Lithuania, Latvia, or Estonia.

Significant quantities of rainbow trout are cultivated in freshwater facilities in Denmark and Germany (over 30 000 tonnes and 22 500 tonnes, respectively, although not all in catchments draining into the Baltic Sea), with lesser amounts in Poland (7500 tonnes), Finland, Sweden, and Estonia. Production of carp in fresh water in Germany, Lithuania, Estonia, and Poland of about 35 000 tonnes is the next most important species quantitatively. Production in fresh water thus greatly exceeds that in salt water in the Baltic region, with Finland being the main producer in marine waters.

Table 14.2.1. Production statistics for aquaculture in the Baltic Sea in 1997.

	Production in tonnes	
Country	**Marine**	**Fresh water**
Denmark	7 000	33 000
Estonia	150	200
Finland	13 000	3 000
Germany	<100	37 000
Latvia	-	300
Lithuania	-	-
Poland	-	29 000
Sweden	2 000	3 000

Use of chemicals and medicine in Baltic aquaculture

In EU countries, medicines for use in aquaculture have to be registered by the appropriate authorities and prescribed by a veterinarian. The medicines used in salmonid fish production are primarily antibiotics and chemotherapeutics to be used in the treatment of bacterial infections. The consumption of antibiotics (oxolinic acid, oxytetracycline, and sulphonamides) in Sweden, Finland, and Denmark is listed in Table 14.2.2, together with the production of salmonids. It should be noted that only a minor portion of Danish fish farms are situated on river systems which discharge into the Baltic Sea area.

Table 14.2.3 demonstrates the sharp reduction in the use of antibiotics in Sweden in the period 1994–1999.

The use of medicines is not considered to be a significant environmental problem in marine aquaculture in the Baltic Sea area. Due to the low salinities, there are no parasite problems that might require the use of antiparasitic agents, in contrast to countries with salmonid production at oceanic salinities. The use of antimicrobial agents in Sweden and Finland has declined in recent years due to increased utilization of vaccines against the most common bacterial diseases, in agreement with the experience in many other countries including Norway, Ireland, and Scotland.

Role of fish culture in eutrophication of the Baltic Sea

The role of fish farming in eutrophication was examined from several perspectives:

a) inputs of nutrients from non-aquaculture sources;

b) calculation of inputs from aquaculture;

c) comparisons of both sources.

A conservative estimate of the amount of nutrients supplied to the Baltic Sea, exclusive of nutrients from fish farming, is given in Table 14.2.4. Data on nitrogen and phosphorus inputs to the Baltic from municipal, river, and industrial sources were supplied by the Helsinki Commission. Data for the input of these nutrients from atmospheric sources were taken from Ackefors and Enell (1990). No data on nutrients transported from the North Sea were included as there is a net outflow of water, extending from the surface to sill depths, from the Baltic Sea except during storm events (Jorgensen and Richardson, 1996).

New technologies and improved feed formulations have permitted a reduction of feed wastage by as much as 70 %. Changes in feed digestibility have permitted a further 35 % reduction in the nitrogen and phosphorus content of the feed, with the consequence that less nitrogen and phosphorus are being excreted per kg of feed eaten.

Over the past decade, total fish culture in the Baltic basin has increased by a modest 14 %. Even with that increase, there has been a dramatic reduction in the amount of nutrients entering the ecosystem from fish farm wastes. The release of nitrogen has been reduced by as much as

Table 14.2.2. Salmonid fish production and quantities of antibiotics used in Baltic aquaculture from 1996 to 1998.

	1996		1997		1998	
	Fish production (tonnes)*	Antibiotics (kg active substance)	Fish production (tonnes)*	Antibiotics (kg active substance)	Fish production (tonnes)*	Antibiotics (kg active substance)
Sweden	6 350	144	5 360	75	6 900	40
Finland	18 000		16 500		16 500	385
Denmark	37 250	508	36 550	995	39 500	825

*Source: Federation of European Aquaculture Producers (FEAP) homepage: www.fishlink.com/feap. The production figures cover both freshwater and marine aquaculture.

Table 14.2.3. Number of farms and quantities of antibiotics used in Sweden.

	1994	1995	1996	1997	1998	1999
Total number of farms	148	118	86	87	32	48
Total amount used in kg	231	232	144	75	40	38

Table 14.2.4. Non-aquacultural nutrient inputs to the Baltic Sea as tonnes per year and as a percentage of the total from each nutrient source.

	Rivers	Municipal	Industrial	Atmospheric	Total
Nitrogen (tonnes yr^{-1})	605 330	60 830	14 580	343 900	1 024 714
	59%	6 %	1 %	34 %	100 %
Phosphorus (tonnes yr^{-1})	40 300	5 220	2 010	6 725	54 274
	74 %	10 %	4 %	12 %	100 %

70 % and the reduction of phosphorus loadings has been in the order of 65 % (Table 14.2.5). Calculated inputs of nitrogen and phosphorus from aquaculture are given in Table 14.2.5 for the year 1997, as compared to values for 1987.

Table 14.2.5. Nutrient inputs to the Baltic Sea from aquaculture.

	Total 1987	**Total 1997**	**1997 Marine**	**1997 Fresh water**
Fish production (tonnes)	133 231	152 277	31 876	107 394
Feed usage (tonnes)	239 816	167 505	35 064	128 873
Nitrogen input (tonnes)	20 336	9 380	1 964	6 014
Phosphorus input (tonnes)	4 077	1 843	386	1 418
N bound in fish (tonnes)	3 997	4 568	956	3 222
P bound in fish (tonnes)	533	609	128	430
Tonnes N in environment	16 339	4 812	1 007	2 792
Tonnes P in environment	3 544	1 233	258	988

The amount of nitrogen entering the Baltic Sea from fish farming in 1997 was something less than half of one percent of the nitrogen entering from all documented sources. At the same time, the amount of phosphorus was less than 2.3 % of the total phosphorus inputs (Table 14.2.6).

Table 14.2.6. Nutrient inputs to the waters of the Baltic and the percent contribution from aquaculture in relation to other sources.

	Total 1997	**1997 Marine**	**1997 Fresh water**
Total N (tonnes)	1 029 452	420 266	608 122
% N from aquaculture	0.47 %	0.24 %	0.46 %
Total P (tonnes)	55 488	14 213	41 288
% P from aquaculture	2.22 %	1.82 %	2.39 %

The above-mentioned values represent nutrient inputs in relation to the total annual supply of nutrients to all aquatic environments in the Baltic Sea. The analysis is further extended to fish farm inputs to the marine environment in relation to other nutrient sources for those specific environments.

No major discrepancies occur when the data are re-analysed habitat by habitat. The dominant portion of fish culture in the Baltic occurs in fresh water. As a proportion of the nutrients supplied directly to Baltic marine waters, marine fish culture again supplies a very small portion of the total nutrient budget. Nitrogen from fish farms accounts for 0.24 % of the nitrogen supplied directly to marine waters and phosphorus from fish farms is 1.82 % of that nutrient supply to marine waters.

Mariculture activities are very unevenly distributed in the Baltic Sea. There does, however, exist the potential for highly localized eutrophication that occurs in the immediate vicinity of a concentration of fish farms.

The ACME concluded, however, that the impact of mariculture on the Baltic Sea can be considered negligible.

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Jørgensen, B.B., and Richardson, K. (Eds.) 1996. Eutrophication in coastal marine ecosystems. Coastal and Estuarine Studies, Vol. 52. American Geophysical Union, Washington, D.C.

15 ISSUES RELATED TO GENETICS

15.1 Genetic Effects of the Release of Cultured Fish in the Baltic Sea

Request

Item 3 of the 2000 requests from the Helsinki Commission: to prepare Chapter 9 on “Marine fish migratory and freshwater species in the Baltic Sea Area” for the Fourth Periodic Assessment of the State of the Baltic Marine Environment, 1994–1998, including effects of rearing upon wild populations.

Source of the information presented

The 2000 reports of the Working Group on the Application of Genetics in Fisheries and Mariculture (WGAGFM) and the Baltic Salmon and Trout Assessment Working Group (WGBAST), and ACME deliberations.

Status/background information

Restocking programmes for Baltic salmon have been conducted by Sweden and Finland for a number of years, particularly in relation to rivers with hydroelectric power plants. These programmes give rise to questions concerning the effect of restocking on the genetic composition of wild Baltic salmon, including:

- What percentage of salmon in an area (or river or group of rivers) could be considered to be wild?
- Is there interbreeding between wild and reared fish?
- What are the implications of the loss or dilution of wild salmon genetic material in the Baltic Sea?

Wild production, its occurrence and relation to fishing regulation

The number of wild Atlantic salmon smolts in the Baltic Sea has increased in recent years as a result of the regulation of fishing and the occurrence of large year classes. In 1997 the number of wild smolts was estimated to be about 0.37 million, approximately 0.48 million in 1998, and in 1999 about 0.6 million. The prediction for 2000 is 1.2 million smolts. In 1998, the proportion of wild smolts to total smolts, including those that were hatchery-reared, was about 7.1 % in the entire Baltic Sea. In 1999, this figure increased to 8.2 %, and it is estimated to increase to about 16 % in 2000.

The spatial distribution of the wild fish depends on the geographical location of their spawning sites and their migratory behaviour. On the basis of estimated production in 1999, the two largest stocks together produce 28 % of the total wild production, and rivers in the Bothnian Bay area account for about 70 % of all wild production.

Two different phylogeographic lineages of Atlantic salmon occur within the Baltic Sea area (Koljonen *et al*., 1999), the older one originating from eastern glacial lake populations, the Ice Lake lineage, and the younger one from Atlantic populations, the Atlantic lineage. Current wild smolt production levels and potential reproduction habitats suggest that the Ice Lake lineage is in greater danger of becoming extinct than is the Atlantic lineage. The populations of the Ice Lake lineage with the oldest and some of the most rare genetic material in the Baltic Sea area are currently to be found in the present salmon stocks of Estonia, Latvia, Lithuania, Russia, and southern Sweden.

In general, the proportion of wild fish is high near the mouths of the wild salmon rivers in early summer and also during the spawning migration in the springtime, usually May, along the Finnish coast, when it can even reach 30 % of total catches (Koljonen and McKinnell, 1996; Koljonen and Pella, 1997). At the national level, local fishing regulations are imposed on coastal fishery and at the mouths of the wild salmon rivers. Finland and Sweden have delayed the opening date of coastal salmon fisheries in the Gulf of Bothnia to restrict the harvest of the early run, when the proportion of wild salmon is at its highest. The provisions of this regulation were made more stringent as of 1996. There are clear indications that the regulation has been effective, in particular by allowing wild fish to escape from the coastal fishery into the spawning rivers (ICES, 1999).

Genetic effects of hatchery rearing and their evaluation

There is some interbreeding between wild and cultured salmon, especially when wild stocks are supported by enhancement releases. In these cases, however, releases are based on river-specific broodstocks to ensure that no non-native genetic material is imported into the wild stocks. The hatchery production is, however, sometimes based on an effective population size that is smaller than that of the wild stock to be supported. The releases as such may then reduce the total genetic diversity of the stocks (Ryman and Laikre, 1991), especially if the number of spawners in the river does not increase due to the releases (Waples and Do, 1994).

The genetic effect of hatchery releases on the scale of the whole Baltic Sea is difficult to evaluate, because very little genetic information is available on the stock(s) before releases or before other human impacts (loss of populations and marked crashes in population sizes of the remaining stocks). Even if it were possible to obtain information on genetic changes in some particular cases and insight into changes in the total structure, their importance for the future evolution of the Baltic salmon remains unknown. Genetic changes due to hatchery rearing may occur in both marker gene frequencies, such as allozyme or microsatellite loci, and in quantitative loci that determine life-history traits such as growth rate and age at maturity or migration behaviour. Estimates of changes in both types of traits exist.

The genetic effects of hatchery releases on naturally reproductive stocks depend on several factors. The genetic change caused by gene flow from a different fish stock depends on the genetic difference between the stocks and on the amount of gene flow. The amount of gene flow to each river depends, not only on the total amount of fish released, but usually on the geographical distance between the release site and the river mouth and also on the release methods. The amount of released salmon smolts in the Baltic Sea is very high, about six million, and at present about 90 % of this production is based on relatively constant hatchery releases. Smolts are produced either from wild spawners caught at river mouths or from captive broodstocks (usually governmental programmes). Broodstock breeding may take place in either long-term or short-term programmes. In long-term captive breeding, genetic material from the original wild stock is no longer available, and the breeding is based on several successive hatchery generations. In short-term breeding, each broodstock generation is renewed, at least partly, with individuals from a wild population, and only second generation hatchery offspring are released.

The majority of the smolt releases are based on water-court decisions and are obligations on hydro power plant companies to compensate for destroyed spawning habitats and lost salmon catches. Straying rates are higher for sea and delayed releases than for river releases. Owing to closure of the rivers, river releases are not always possible.

The genetic difference between hatchery and wild fish may be caused either by different genetic origin or by hatchery rearing processes (non-random sampling of spawners or selective changes in rearing practices). The genetic effects of hatchery rearing on Baltic salmon stocks have been studied by comparing the genetic differentiation pattern and characteristics of stocks and stock groups of wild and hatchery origin and also the amount of diversity (mean heterozygosity or number of alleles) between the wild and hatchery derivatives of the same river stock. In addition, crossing experiments have been conducted to estimate the genetic change in quantitative traits caused by selective forces in hatchery rearing.

Changes in marker gene frequencies

An allozyme study of Baltic salmon stocks suggests that releases of hatchery fish and the loss of several original stocks have caused loss of the isolation-by-distance differentiation pattern that was originally present in the genetic differentiation of the Baltic salmon stocks. Thus, hatchery rearing has caused allele frequency shifts and random changes in historical genetic differences between the stocks. The genetic diversity of hatchery stocks is somewhat lower than that of wild stocks, and the wild stocks are more different from each other than are the hatchery stocks. This is probably due to the fact that the population sizes of hatchery stocks tend to be smaller than those of wild stocks.

Changes in quantitative traits

A crossing experiment in which the offspring of wild and reared parents from the same river stocks were compared revealed some changes in quantitative traits (i.e., traits under both environmental and genetic influence, usually involving several loci). Quantitative genetic traits are directly related to the viability and fitness of the stocks, contrary to the genetic allozyme variation. The growth rate of the offspring of hatchery parents was statistically significantly higher than that of the offspring of wild parents when they were smolts and also later in the sea. The growth rate of the hybrid group with wild and reared parents showed intermediate capacity (Kallio-Nyberg and Koljonen, 1997).

The same crossing experiment showed that the age at maturity of the offspring of the reared parents was lower than that of the offspring of the wild parents. Especially the proportion of mature one-sea-year old fish, mostly males, was higher in the hatchery group (52 %) than in the wild group (34 %).

Hatchery rearing may include selective factors that might change the genetic composition of quantitative traits. To what extent this has happened is unknown. In the case studied, selection had not been intentionally avoided and the collection of spawners had led to overrepresentation of larger fish in the broodstock, which in turn caused a decrease in average age at maturity and an increase in the proportion of one-year old mature fish.

Conclusions

- Evidence exists that some changes, albeit not substantial, have occurred in both the diversity levels of the marker genes and the inherited life-history traits of Baltic salmon.
- Some genetic changes will be inevitable in the future, too, because artificial reproduction can never totally mimic the genetic diversity in the wild stocks.
- There is no return to the original state of the Baltic salmon populations, and conservation of genetic diversity should thus be planned onwards from the present situation.

Need for further research or additional data

The genetic effect of hatchery releases on the scale of the whole Baltic Sea is difficult to evaluate because very little basic information is available. To obtain such information, monitoring of genetic changes (at least of diversity levels of marker genes) should be conducted. Hatchery rearing may include selective factors that might result in changes in the viability (fitness) of the reared populations in the wild environment. However, knowledge within this field is limited.

Recommendations

ICES recommends the following:

1) For maintaining genetic diversity, many populations are required and each must be abundant enough to protect genetic diversity. Thus, it is important to conserve areas where substantial natural reproduction can still take place and control sources of mortality wherever possible. The conservation of these areas should be prioritized.
2) To retain the larger-scale genetic structure, major groupings of populations need to be taken into account. Thus, separate strategies are needed for the Ice Lake and Atlantic lineages of salmon within the Baltic Sea.
3) Stock transfers between the ranges of the Ice Lake and Atlantic lineages of salmon should be strictly avoided.
4) The ranges (distances) of stock transfers within the lineages should be minimized.
5) Activities causing straying, such as delayed releases and sea releases, should be minimized.

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15.2 Potential Genetic Implications of Endocrine Disruption

Request

This is part of continuing ICES work on the effects of marine contaminants on biota.

Source of the information presented

The 2000 report of the Working Group on the Application of Genetics in Fisheries and Mariculture (WGAGFM) and ACME deliberations.

Status/background information

Endocrine disruptors (EDs) are chemical substances interacting with hormone regulatory systems of aquatic organisms, or acting as hormones themselves. Most of the current research activities on EDs concentrate on the identification of EDs and the magnitude of their activity in certain species. There is a considerable body of literature on the effects of EDs on aquatic organisms, with the most frequently observed effect being feminization. The opposite effect, i.e., masculinization, and thyroid-like activity have also been observed.

Genetic effects of EDs have been observed in a variety of individual organisms other than fish, but there is no reason to assume that the same effects could not occur in fish. The effects of oestrogen-like endocrine-disrupting chemicals on the individual genome have been summarized by Roy *et al.* (1998). Diethylstilbestrol (DES), a synthetic oestrogen, has been demonstrated to alter the genome via numerical chromosome changes (aneuploidy). DES is also capable of creating instability by producing mutational changes in both the mitochondrial and nuclear genome. Chromatid and chromosome breaks have also been shown to be induced by EDs.

At the population level, the effects of EDs in fish are not well studied or understood. Fairchild *et al.* (1999) have suggested that declines of Atlantic salmon populations could be due to EDs, but no specific mechanisms have been proposed. However, in general, it is an issue of concern that a change in sex ratio or reduced reproductive success will lead to a decline in effective and absolute population size and, consequently, to a reduction of genetic variability.

Most of the findings on the effects of EDs are from laboratory or freshwater environments. Feminization or masculinization and complete sex reversal to functional male or female organisms are likely to be rare in the marine environment due to the lower concentrations of EDs compared to concentrations found in rivers and used in laboratory experiments. Contaminated estuaries, however, may be an exception to this (Allen *et al.*, 1999a, 1999b; Matthiessen *et al.*, 1998). Several examples of intersex (i.e., the development of both male and female gonad tissue in male gonads) have been described in flounder (*Platichthys flesus*) caught in Liverpool Bay, the Mersey Estuary, and the Tyne Estuary, UK. It should be noted that all field observations to date have merely recorded the phenotypic sex of fish exposed to oestrogenic endocrine disruptors. It is possible, however, that some individuals recorded as phenotypic females may actually be genetic males which were completely feminized during larval development. This issue can only be decided when molecular probes for determining the genetic sex of fish become more readily available. However, abnormal sex ratios in wild fish have not been widely observed to date.

Need for further research or additional data

Very little is known about the potential effects of endocrine disrupting substances on individuals and populations. More research concerning the natural underlying sex-determining mechanisms is needed before the effects of endocrine disruptors on the phenotypic sex of finfish and/or shellfish can be determined.

Taking note of the information above, the ACME recommended that, in those cases where endocrine disruption has been demonstrated in individuals in the wild, genetic analysis of exposed populations should be considered in order to gain evidence for the putative loss of genetic diversity.

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15.3 Potential Genetic Implications of Commercial Fisheries on Deep-Sea Fish Stocks

Request

This is part of continuing ICES work on genetic studies of fish.

Source of the information presented

The 2000 report of the Working Group on the Application of Genetics in Fisheries and Mariculture (WGAGFM) and ACME deliberations.

Status/background information

There are a number of deep-water fish stocks that are presently fished or have the potential of being fished. In addition, there are stocks fished on the continental shelf but whose distribution extends into deeper water.

Although there have been genetic studies conducted on certain deep-water fish stocks, with additional studies now in progress, little is known about the potential effects of fisheries on these species.

As catches of traditionally exploited marine resources are not likely to increase, the possibility of harvesting other species on new fishing grounds should be investigated intensively in the future. Deep-water species constitute a plausible alternative, with many species already being exploited, either directly targeted or as a by-catch product. However, it is known that some of the species exhibit very slow growth and reach sexual maturity at a late age. This is likely to affect their reproductive output and potentially makes them vulnerable to extensive harvesting. Furthermore, it is known that the distribution of some of these deep-water species extends into international waters and they are, therefore, subject to uncontrolled fishing. Consequently, there is a strong need to gather basic data on the population dynamics and genetic population structures of these species in order to evaluate the potential effects of fisheries on them.

Population genetics research is being carried out on some deep-sea species, e.g., *Sebastes* spp., on both sides of the Atlantic. However, the lack of general biological information and knowledge of the genetic structure of deep-water fish populations is recognized.

Need for further research or additional data

Noting the above information, the ACME recommended that:

1) high priority be given to research aimed at deep-water fish species, especially with regard to their vulnerability to overexploitation and the role of specific habitats such as sea mounts;

2) the research efforts at this stage should concentrate on a few species, so that more extensive biological data (general biological features, population dynamics, population genetics) can be obtained about them. These species could then serve as model species, both in order to assess the importance of specific biological features of deep-water fish (such as slow growth) in relation to harvesting and potential depletion of genetic resources, and in order to be able to focus later research activities on other deep-water species.

Request

This is part of continuing ICES work on ecosystem effects of marine aggregate extraction.

Source of the information presented

The 2000 report of the Working Group on the Effects of Extraction of Marine Sediments on the Marine Ecosystem (WGEXT), the report "Effects of Extraction of Marine Sediments on the Marine Ecosystem" to be published in the *ICES Cooperative Research Report* series, and ACME deliberations.

Status/background information

The ACME reviewed and accepted several sections of the report of the Working Group on the Effects of Extraction of Marine Sediments on the Marine Ecosystem (WGEXT) containing information and discussions on effects of marine sand and gravel extraction on marine ecosystems, including quantities of material extracted, impacts on biota, and the effect of turbidity caused by dredging. Approaches to environmental impact assessment were also reviewed. This material is summarized below.

The ACME noted that WGEXT has discussed the updating of the 1992 Code of Practice for the Commercial Extraction of Marine Sediments; a summary of good practice will be considered at the 2001 meeting of WGEXT.

Activities have also been initiated on the further development of guidelines for environmental impact assessments of marine aggregate extraction, and it is intended that a draft set of guidelines will be produced in 2001.

16.1 Current Marine Extraction Activities and Results of Assessment of their Environmental Effects

The ACME took note of the status of marine extraction and dredging activities in ICES Member Countries, as reported to WGEXT. Particular emphasis was given to a review of approaches to environmental impact assessment and related environmental research, as summarized in the paragraphs below.

Canada

Although no marine mining presently takes place in Canadian waters, harbour dredging and port maintenance are major activities. Cooperative projects to assess the effects of seabed trawling and clam dredging on seabed habitats are in the final stages of field survey and the results can apply to potential seabed mining activities. These projects are being conducted together with the Department of Fisheries and Oceans. The Geological Survey of Canada is providing geoscience survey and interpretative support. Results of these surveys will be a quantitative assessment of the effects of the bottom fishing gear on seabed alteration, biodiversity, community complexity, and ecosystem reoccupation. Research planned for the future will assess the effects of scallop fishing on the seabed.

Denmark

<u>Øresund Link</u>

All primary dredging works related to the project have been completed. The removal of temporary structures continues, but all marine works are scheduled to be finished before the opening of the Link on 1 July 2000. A total of 14.2×10^6 m^3 of material has been dredged during the project, with an average spill of 4.1 %.

Only minor effects have resulted from the construction works. These effects were predicted and are within the tolerance limits for environmental effects required by the authorities. The sand fill for the construction of the Link was dredged on Kriegers Flak in the Baltic Sea. Before the commencement of the project, a detailed resource assessment and an environmental impact assessment (EIA) of sandfill extraction was carried out on Kriegers Flak in the Baltic by the Øresund Consortium. The assessment was prepared in accordance with the EC Directive 85/337.

During the project, 1.3×10^6 m^3 of sand fill was dredged from Kriegers Flak, with a spill rate of 2.8 %. Sediment spill and the environmental impact of the dredging activities were monitored in detail during the dredging operation, and a final report on the findings is scheduled for completion in April 2000. The preliminary results indicate, in accordance with the EIA, that there is no environmental impact outside 1000 m from the dredging area.

<u>Århus Harbour</u>

Preliminary results from the monitoring of the bottom fauna after dredging of approximately 6×10^6 m^3 of sand indicate that the changes outside the dredging areas are very small and of the same magnitude as in the reference area. The results in both the impact area and the reference area indicate a significant and parallel increase in the number of species and the abundance.

<u>Eastern Kattegat</u>

Investigations of the potential impact on bottom fauna and birds are being carried out off Limfjorden prior to the commencement of dredging of sand for cement production.

Stigsnæs

An Environmental Impact Assessment in accordance with the EC Directive 85/337 has recently been conducted for a proposal to construct a Container Terminal Hub near Stigsnæs, western Zealand. The public procedure is scheduled to run from April to July 2000. The project includes the dredging of 5.6×10^6 m^3 of sand fill in a very environmentally sensitive area. To fulfill the environmental requirements, direct pumping from the dredging site and the use of sedimentation basins are expected to be necessary.

North Sea

The Danish Coastal Authority (DCA) is launching an Environmental Impact Assessment in accordance with the EC Directive 85/337 for areas in the North Sea to be used for the dredging of sand for beach nourishment in the next ten years. The project will include geological and biological mapping. In addition, DCA is planning to carry out a separate Environmental Impact Assessment for both onshore and nearshore nourishment.

Impact of dredge spill on the benthos

The Danish Forest and Nature Agency has initiated a research project on the impact of dredge spill on benthos, in cooperation with the National Environmental Research Institute. A detailed study of the ecological consequences of dredging in coarse sediments was started in May 1996. Especially the effects on the benthic flora and fauna on surrounding stone reefs will be evaluated.

Environmental effects of dredging in the North Sea

The Forest and Nature Agency and the Coastal Protection Agency have initiated a monitoring programme off the west coast of Jutland to study the effects of the dredging of sand for beach protection. The study is based on a comparison with simultaneous changes in a reference area. The post-nourish temporal development is analysed using the BACI concept (B(before) A(after) C(comparison) I(impact)). The study showed that a complete quantitative recovery including the number of species, the abundance, and the biomass of the bottom fauna had occurred in less than one year after the sand extraction. However, the predominance of a supposed opportunistic species of polychaete (*Spio filicornis*) in the borrow area may indicate a pioneer recolonization. The impact of sand extraction on the predator populations is limited due to a patchy exploitation pattern leaving plenty of food in 70 % of the undisturbed bottom and a recovery of the benthic biomass in less than one year.

Effects of excavations for natural resources on benthic epifauna

The purpose of this project is to determine different kinds of effects on benthic epifauna as a result of excavations for sand, gravel or stones. The aim is to describe both short-term and long-term effects related to spill from the dredging operations. The results of the investigations will be combined with data on spill percentage, particle size distribution, and sedimentation in order to obtain an overall estimate of secondary effects of these operations.

Effects of sediment spill were evaluated on three levels: short-term effects, long-term effects, and worst case scenario. Short-term effects of elevated levels of suspended material were examined by measuring filtration rates and activity patterns in four or five different phyla of epibenthic suspension feeders, i.e., ascidians, bryozoans, bivalves, polychaetes, and marine sponges. Performing growth experiments at elevated concentrations of inorganic suspended material on two or three different model organisms will elucidate long-term effects. In the worst case scenario, experimental plots in the field were covered with inorganic sediment and divers will follow the recolonization of the plots for one year, using direct observations.

Preliminary results show a reduced filtration rate in some of the species investigated. This was expected since most of the species are not capable of sorting particles. The reduced food intake is due to dilution of food with inorganic particles derived from sediment overspill. However, the results were not unequivocal since some species, such as bryozoans, did not show changes in feeding activity. In the worst case scenario, the first visit after the sedimentation event showed a marked change in species composition and abundance in the experimental plots. Final results are expected to be ready by the end of 2000.

France

A study on sedimentological and biological effects of sand deposition was carried out around the extraction site of Dieppe in 1999. A negative impact, including an increase in fine sand content and a decrease in species richness, abundance and biomass, was observed in the eastern areas of the study site, in line with the direction of the prevailing tidal currents. In addition, a positive effect was detected northwards and westwards, mainly as an increase in biomass, although no effect in species richness or abundance could be detected. Comparison with Poiner and Kennedy's (1984) results in Australia shows that macrobenthic communities are mainly affected by a change in the sediment type.

Germany

Regeneration of sediment extraction sites in the North Sea and Baltic Sea

In March 1999, the Federal Maritime and Hydrographic Agency (BSH) initiated a three-year research project to investigate the sedimentological processes which control the refilling of sediment extraction sites along the North Sea and Baltic Sea coasts. The extraction sites studied include: Westerland II (North Sea), Pohnshalligkoog near the island of Pellworm (North Frisian Wadden Sea), Graal-Müritz 1 and Tromper Wiek (both in the Baltic Sea off Mecklenburg-Vorpommern).

Westerland II is a sand extraction site, which is used for coastal protection of the island of Sylt. The Tertiary sands and gravels are extracted by anchor dredging from 10 m to 12 m deep pits in a water depth of 15 m. Since extraction commenced in 1984, the site has been frequently echo-sounded and these data are used to evaluate the behaviour of the adjacent sea floor. The pits act as sediment traps for fine-grained material, which accumulates in the deeper parts. Analyses of organic pollutants and the heavy metal composition of the muddy refill will be used to indicate the source of the material (i.e., whether it originates from the Wadden Sea or from riverine input).

A single sediment extraction exercise was conducted in 1994 at Pohnshalligkoog near Pellworm for a dike protection in 1 m water depth. The pit was stable for several years and in 1999 was almost completely refilled. Sediment cores are scheduled to be retrieved in summer 2000 to obtain information on the sedimentological processes that are responsible for the refilling of pits in the Wadden Sea.

There is a very large activity along the Baltic Sea coast of Mecklenburg-Vorpommern to extract sand and gravels from shallow waters for coastal protection and industrial use, respectively. At Graal-Müritz 1, sand extraction is taking place in 2000 for the protection of a nearby coast. Trailer dredgers will extract about 500 000 m^3 of mobile sands from an area that is characterized by a sand thickness of 2 m to 4 m. Since the beginning of the project, the sediment distribution has been mapped three times in order to compare the natural variability in this area with anthropogenically induced processes responsible for refilling the furrows.

In the Tromper Wiek, both gravelly sands for industrial use and sands for coastal protection are extracted from water depths of 10 m to 20 m. Gravelly sands are sieved on board the anchor dredgers and the sandy fraction is returned to the sea with the overflow. The behaviour of this mobile sediment and the stability of the pits are of major interest at this site.

Ireland

Locations and extent of the main herring (*Clupea harengus*) spawning grounds around the Irish coast

This project was funded under the Marine Research Measure of the Operational Programme for Fisheries (1994–1999), administered by the Marine Institute, and partly funded by the European Regional Development Fund. The aim of this research was to delineate the locations and extent of the major herring (*Clupea harengus*) spawning grounds around the Irish coast and, where possible, to map the extent of the actual spawning beds within these areas. This project was undertaken in response to a growing interest in coastal and offshore resources, resulting, for example, in applications for sand and gravel extraction and the dumping of dredge spoil from maintenance dredging. The information obtained was used to provide an atlas of the areas to which careful consideration should be given before issuing licenses for the extraction of marine aggregates and for dumping at sea.

It was found that preliminary surveys of the spawning grounds using the RoxAnn™ seabed discrimination system were useful in identifying areas containing suitable spawning substrates prior to actually surveying for herring eggs. This identified areas where herring are likely to spawn and also reduced the size of survey areas by providing information on the location of substrates that would be considered unsuitable for spawning (i.e., mud and fine sand). These data were also used as a method of identifying potential herring spawning grounds.

This study was undertaken in order to investigate the potential threat posed by this operation to one of the more important south coast herring spawning grounds, which is situated adjacent to the dumpsite for dredge spoils. Another RoxAnn™ survey was carried out to map the seabed sediments within a proposed aggregate extraction site that was located within a well-known spawning ground off the southeastern coast of Ireland. The report concluded that the full extent of the spawning grounds should be protected in line with the precautionary approach to fisheries management.

The Netherlands

EIA of an area off the coast of Zuid-Holland

An Environmental Impact Assessment has been carried out within an area off the coast of the province of Zuid-Holland. This study is general in nature and has focused on the environmental impact of the extraction of sand for concrete from a depth of 5 m to 30 m below the seabed. Special attention was given to the way the cover layer of lower quality sand will be handled. The results of the environmental impact assessment point towards a refill of the pit with the sand from the cover layer, or to a combination with land-reclamation projects.

United Kingdom

Assessment of rehabilitation of the seabed following marine aggregate dredging

The Centre for Environment, Fisheries and Aquaculture Science (CEFAS) in collaboration with Hydraulics Research Wallingford and the British Geological Survey have been commissioned to undertake a four-year study to assess the rehabilitation of the seabed following marine aggregate dredging. The project is jointly funded by the Department of the Environment, Transport and the Regions (DETR), the Ministry of Agriculture, Fisheries and Food (MAFF), and the Crown Estate.

This field-based study is designed to enhance understanding of the processes leading to physical and biological recovery of the seabed following dredging, thereby aiding the identification of practices to minimize environmental harm at such sites, and to promote rehabilitation on cessation. In combination with ongoing CEFAS research and development, which ranges from an evaluation of the cumulative effects of dredging activity and the modelling of plume dispersion, to studies of environmental recovery following experimental dredging and the development of new methodology for characterizing gravel habitats, the outcome will provide a significant advance in understanding, and hence in the scope for effective management, of commercial extraction and its aftermath.

Procedural guidelines for the conduct of benthic studies at aggregate dredging sites

DETR has commissioned CEFAS to prepare guidance on procedures for undertaking benthic studies in relation to aggregate dredging activities. Its purpose will be to provide a more consistent approach to such assessments, and to enhance the quality and compatibility of the data generated. The output will be relevant either to studies undertaken as part of a pre-application Environmental Impact Assessment, or as part of a programme to monitor the effects of dredging.

United States of America

Borrow area monitoring off New York and New Jersey

Organizations undertaking the research project are the U.S. Army Corps of Engineers and various sub-contractors. The funding body is the U.S. Army Corps of Engineers.

The project studies active borrow areas off the coast of New Jersey for benthic resources (primarily shellfish), benthic prey for demersal fish, and finfish populations. Unpublished reports include the 1999 Bio Monitoring Program report Phase II–III, and Raritan Bay and Sandy Hook Bay Offshore Borrow Site Area Analysis, and the 1999 Finfish and Invertebrate Summary Report.

Monitoring at excavated borrow sites showed decreases in both total abundance and biomass of benthic communities of the dredged borrow area, but both appeared to recover within 8 to 9 months. Biomass composition was also impacted, but had not completely recovered one year after dredging. Likewise, the average weight of sand dollars, the dominant biomass, was still lower at dredged sites than at reference sites one year after dredging.

Similar studies have been conducted or are under way in borrow areas along the New York ocean coastline.

16.2 Biological Effects of Dredging in the Marine Environment

Impact of turbidity plumes

A factor that may have significance for the benthos in the vicinity of dredging is that of sediment resuspension. The draghead itself can agitate the sediment causing a noticeable increase in the amount of nearbed suspended solids. However, the outwash from spillways on the dredger generates a far greater quantity of suspended material (Moran, 1991).

The morphology and behaviour of fine sediment plumes, under varying hydrodynamic regimes, have been investigated by Pennekamp and Quaak (1990). Historically, however, there are fewer references to similar studies involving the dredging of non-cohesive material; only very recently have similar studies been performed in connection with major dredging projects such as those currently under way in Hong Kong (Hitchcock and Drucker, 1996), off the east coast of England, and between Denmark and Sweden (Øresundskonsortiet, 1998a, 1998b).

Plumes of suspended material can arise from three distinct sources, namely: 1) the mechanical disturbance of seabed sediments by the draghead; 2) overspill of surplus sediment/water mixture from the vessel hopper; and 3) the rejection of unwanted sediment fractions by screening.

Plumes associated with the latter two sources have been termed surface plumes and their volume and duration are linked to the particle size of the sediment (mud content) and to the local hydrodynamics (wave and tidal actions).

A recently commissioned study in the UK indicated that the “bulk” of a dredge plume (approximately 80 % of the total discharged sediment by weight is composed of sand-sized particles) collapses to the seabed within a few hundred metres of the dredger. This observation was also made in a recent study of aggregate plume dynamics in the English Channel by Hitchcock and Drucker (1996). They were able to demonstrate that suspended sediment (> 0.063 mm) decayed to background levels over a distance of 200 m to 500 m from the point of release. Nevertheless, the remaining 20 % of sediment in the plume will largely be composed of particles < 0.063 mm in diameter and this fraction will potentially be dispersed over much greater distances due to their very low settling velocities. Similar work in Dieppe has confirmed these findings (Desprez, 1997).

The impact of increased sediment resuspension caused by dredging in deposits of clean mobile sands or in areas with high natural background levels of turbidity, such as at the mouth of estuaries, or in high-energy areas close to eroding coastlines such as parts of the North Sea and the Bay of Fundy, is arguably of less concern because of naturally high loads of suspended sediment caused by tides and wave action in these areas (Millner *et al.*, 1977). Recent results from analyses of sediment samples taken from Danish marine aggregate areas have shown that the content of fines exceeds 5 % only in a few samples (Nielsen, 1997). Dredging mainly for sand in Dutch coastal waters is expected to cause a maximum turbidity plume of 32 mg l^{-1}, during slack water conditions. However, turbidity levels in excess of this figure, for the same area, were measured during storm conditions.

Effects on seabed communities

The ACME noted that there are a number of effects that plumes of suspended sediments potentially can have on the marine environment. The level of knowledge in relation to some of the main effects is summarized below.

- Increased sediment deposition and the modification of sediment type are potentially the most significant effects on marine ecology after the direct removal of the substratum and associated communities by the drag pipe.
- There is little field evidence regarding the level and nature of the impact.
- In sheltered areas, sedimentation is mainly confined to a zone of a few hundred metres from the point of discharge (e.g., Newell *et al.,* 1999).
- In high-energy environments, impacts of deposition can extend far from the dredging area (> 1 km). If significant deposition of sand occurs in areas that have a similar sediment type, the impacts on the benthic communities are likely to be small or can actually result in an enhanced biomass or diversity.
- The effects of elevated turbidity and sedimentation are more significant in environments that have naturally low concentrations of fine sediments, particularly in areas dominated by gravel substrata.
- Reduced light penetration can affect macrophytes (see, for example, ICES (1997) and the 1998 reports on the Øresund fixed link (Øresundskonsortiet, 1998a, 1998b)).

Effects on fish

- Fish are potentially sensitive to sediment in the water column and material settled on the seabed.
- Dredging plumes may affect fish larvae and eggs that concentrate in the surface layers. However, there has been little work on the subject.
- Fish spawning is affected where the settlement of sediment directly smothers fish eggs or changes the composition of the substratum of a nursery area.
- An increase in the fines content may prevent the eggs of species such as herring and sandeel from adhering to the sediment, and can cause smothering of the eggs.
- Demersal spawning species which require a particular sediment type on which to spawn are possibly the most susceptible to changes in sediment composition.
- Possible direct effects include abrasion resulting in the removal of mucus and the clogging of gills. However, it is uncertain whether these effects are likely to occur to a significant extent in practice, as fish are likely to avoid areas with sufficiently high turbidity.
- Plumes may change the behaviour of fish that target their prey visually. Fish that normally inhabit relatively clear water may have their feeding pattern impaired.
- The potential for impacts on fish migration depends on the duration and timing of dredging.
- Migration is seasonal, so the impact would be more significant if the dredging occurred during the critical period.
- Eggs and larvae of fish may be impacted by the release of toxic chemical compounds from sediments during dredging.

Effects on shellfish

- Settlement of the sediment can affect epifaunal crustaceans.
- A localized redistribution of crustaceans can occur as a reaction to habitat loss.
- Silting up of pots can lead to declining catches from traditional fishing grounds.
- The limited near-field dispersion and short-term duration of significant sediment plumes would only result in localized direct impacts on shellfish. However, such effects can impact on commercial shellfisheries.

- Eggs and larvae of shellfish may be impacted by the release of toxic chemical compounds from sediments during dredging.

Effects on primary production

- Increased turbidity may also depress phytoplankton productivity due to a reduction in light penetration into the water column. This effect is localized and is unlikely to be significant in terms of total production.

Cumulative effects

- In the context of sediment plumes, a cumulative impact is considered to constitute more than one dredging event creating multiple plumes.
- In isolation, one dredging regime may not have a significant impact. However, the cumulative effect of several dredging regimes might be to exceed a critical threshold.

Nature of the biological effects

The most obvious impact of sand and gravel extraction is the removal of the substrate and the resulting destruction of the benthic biota. However, not all of the benthos will be retained in the hopper; for example, it was observed during a study of dredger spillway outwash contents that a significant quantity of the dredged benthic fauna was returned back to sea (Lees *et al*., 1990). Nevertheless, the possibility of any dredged fauna surviving the extraction process and re-establishing itself on the seabed is very unlikely, although such fauna will probably provide a very valuable source of food for scavenging benthic invertebrates and fish.

The Dutch studies of van Moorsel and Waardenburg (1990, 1991) and van Moorsel (1993) showed that, soon after dredging, a reduction occurred in the abundance (70 %), biomass (80 %) and, to a lesser extent, the number of species (30 %). The large reduction in biomass was attributed to a loss of large molluscs, particularly *Arctica islandica* and *Dosinia exoleta*. They concluded that while the densities and number of species returned to pre-dredged levels within one year, the biomass had not recovered some two years later. These findings were supported by work carried out in the UK which examined the biological and physical responses following a controlled dredging event off North Norfolk (Kenny and Rees, 1994, 1996). The trends observed for the densities and numbers of species during the above studies are also supported by research carried out elsewhere. For example, in areas where sediment transport was high, due to strong tidal currents or wave action, the community had recovered within one year of dredging (Johnson and Nelson, 1985; Pagliai *et al.,* 1985).

Results obtained by Johnson and Nelson (1985) and van Dolah *et al.* (1984) showed that within a few weeks of dredging, significant reductions in animal densities were observed, although the numbers of species were unaffected. A possible explanation for the latter observation was provided by Hall *et al.* (1991), who suggested that a likely pathway for early post-dredging recolonization was by passive translocation of animals during storms. Indeed, the presence of adult infauna in a channel within hours after dredging was noted by van Dolah *et al*. (1984). It was concluded that their presence was due to sediment sliding down the walls of the dredged channels from nearby unaffected areas.

It seems likely that many species are capable of being naturally transported by small-scale sediment disturbances (Rees *et al.,* 1977; Hall *et al.*, 1991; van der Veer *et al.*, 1985; van Dolah *et al.*, 1984). However, large-scale disturbances are most likely to be species-selective (Rees *et al.*, 1977). These disturbances could involve a large amount of sediment being transported for a short time (storms) or, indeed, a smaller amount of sediment being transported over longer periods (tides). Both types of disturbance may result in a reduced and specialized fauna more resilient than a diverse or “biologically accommodated” fauna (Boesch and Rosenberg, 1981). Species characteristic of a community subjected to large-scale physical disturbances are likely to utilize passive translocation as a major strategy for colonization of new habitats (van der Veer *et al.*, 1985).

Maurer *et al.* (1981a, 1981b) carried out experiments on the lethality of sediment overburden on selected macro-invertebrates. They concluded that many motile epibenthic and infaunal animals could withstand a light overburden of sediment (about 1 cm), especially when the overlying sediment was native to their habitat. They also found that increased depth and frequency of burial caused greater mortality, a finding confirmed in Dieppe studies (Desprez, 1997). In addition, mortality was linked to water temperature, such that mortality was greater during the summer months than in the winter.

Another factor which may have significance for the benthos in the vicinity of dredging is that of sediment resuspension. The draghead itself can agitate the sediment causing a noticeable increase in the amount of nearbed suspended solids. However, the outwash from spillways on the dredger generates a far greater quantity of suspended material (Moran, 1991). The effect of suspended inorganic particles on aquatic marine life has been reviewed by Moore (1977) and Newcombe and MacDonald (1991). Marine benthic invertebrates vary greatly in their tolerance to the amount and type of suspended solids (Newcombe and MacDonald, 1991), but there seems to be some correlation between the normal habitat of the species and its sensitivity to suspended particles (McFarland and Peddicord, 1980).

Hard-bottom communities tend to be dominated by epibenthic grazers and suspension feeders which are generally more sensitive to excessive amounts of suspended sediment (McFarland and Peddicord, 1980). Suspension-feeding organisms may be stressed by the abrasive effects of sediment passing over their feeding and respiratory structures. In addition, surface grazers and deposit-feeders may be sensitive to changes in the composition of the microscopic fauna found on the surface of stones and shells (e.g., diatoms, bacteria, and protozoa commonly known as the "microbial film") since this often constitutes a major part of their diet (Turner and Todd, 1991).

In Denmark for the Øresund project, the total amount of spill was limited to 5 % of the total dredged sediment. The EIA model predictions indicated a decrease in eelgrass due to shading from suspended material of around 25 %. Subsequent monitoring showed an actual decline of around 15 % (Øresundskonsortiet, 1998a, 1998b). In the Spanish sub-project of RIACON at the Costa Durada in the Mediterranean Sea, special attention was given to the *Posidonia* (sea grass) community which formerly extended down to 23 m to 24 m, but which now has a narrower depth range. The authors of the report of the study concluded that there is no risk to the sea grass due to dredging activities in the Mediterranean as long as a minimum distance of 1 to 2 km is maintained (Essink, 1998).

Apart from the impact on adult organisms, a change in the biological and physical composition of the sediment may hinder the settlement of benthic larvae. It has been shown that the presence of a microbial film induced the settlement of certain sessile invertebrates such as barnacles (Rodriguez *et al.,* 1993). Any change in the composition or biomass of the film may therefore reduce the potential for settlement. In addition, it has been shown that bed roughness, and composition and texture of the sediment are important factors for larval settlement (Crisp and Barnes, 1954), all of which may be affected by dredging, which in turn may also alter the types of species which colonize the area.

Summary of impacts and ecological response

A model of macrobenthic community response to the effects of dredging is now emerging. The response may be divided into three phases, namely:

Phase I comprises an initial recolonization by the dominant taxa present before dredging. These animals are predominantly opportunistic in behaviour and they significantly contribute to an increase in the overall abundance and total numbers of species during the first few months following the cessation of dredging.

Phase II is characterized by a low community biomass which may persist for several years. This may be caused by increased amounts of sediment (mainly sand) in transport which is also responsible for the erosion of dredge tracks, infilling of dredge pits, and the scouring of the epibenthos. In time (after about two years at the Norfolk site), the sand transport reaches the pre-dredged equilibrium state, which results in Phase III.

Phase III of the recovery is characterized by a significant increase in the community biomass.

Clearly, the same biological and physical responses to dredging as observed in the above studies cannot be assumed to occur elsewhere, i.e., the findings are site-specific. Nevertheless, it may be concluded that dredging of "commercial" sand and gravel deposits in areas of relatively high natural disturbance, e.g., off the east coast of England (Kenny *et al*., 1991) and off the Dieppe in the English Channel (Desprez, 1997), may be of little long-term (i.e., three years) biological "significance" due to the potential speed of physical and biological recovery following dredging. This conclusion clearly has wider implications for the environmental assessment of aggregate dredging, and therefore requires further validation by quantitative field sampling at these and other locations.

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Request

Habitat classification is a new field of interest for ICES.

Source of the information presented

The 2000 reports of the Study Group on Marine Habitat Mapping (SGMHM) and the Working Group on the Effects of Extraction of Marine Sediments on the Marine Ecosystem (WGEXT), and ACME deliberations.

Status/background information

As part of its adopted scientific objectives, ICES aims to develop a classification system and to map coastal marine habitats, continental shelves and slopes, and the open ocean. In order to define the intrinsic value of a habitat and its management requirements, a classification system must first be established for the ICES area, followed by subsequent habitat mapping.

Increasing human impacts such as fishing, shipping, land reclamation, and oil drilling have resulted in environmental pressure on marine areas. The international community has reacted by committing itself to a number of international agreements (e.g., OSPAR Convention of 1992, Annex V) specifying that a precautionary approach be adopted to prevent areas from suffering irreversible ecological damage. As human-induced changes of the marine environment are known to have potentially large-scale impacts, it is important to develop a classification system that is valid on an international scale. ICES will need to cooperate with its partner organizations (e.g., OSPAR, and the European Environment Agency (EEA)), which are also active in this field. In this section, recent developments in marine habitat classification and mapping are discussed as part of the ongoing work of ICES and its partner organizations in this field.

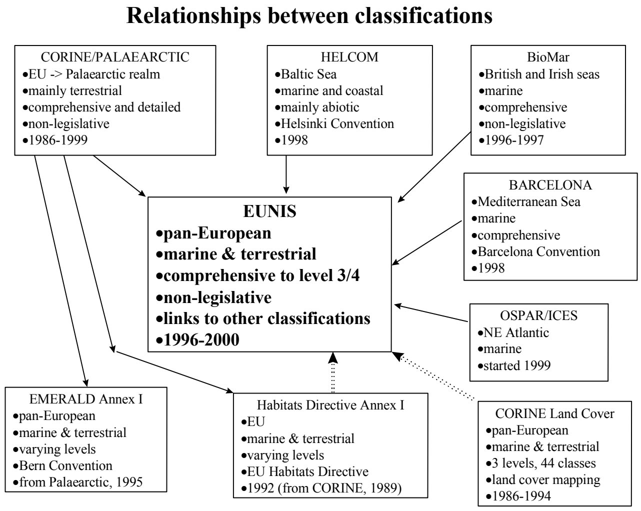
Developments in Marine Habitat Classification

In different parts of the world, initiatives are being taken to develop habitat classification systems as a common language for those who are involved in protecting and conserving threatened ecosystems. Two of these initiatives are described here: (1) the EUNIS classification system, and (2) the ARC system.

(1) EUNIS

The EUNIS habitat classification has been developed at the initiative of the European Environment Agency (EEA). The EUNIS report was recently finalized (Davies and Moss, 1999); the classification is now open for comments and feedback from the public. EUNIS stands for “EUropean Nature Information System” and it aims to provide a common framework for a European habitat classification, for terrestrial as well as aquatic ecosystems. This classification builds on the CORINE/Palaearctic classification. The classification of marine habitats is largely derived from the BioMar project (Connor *et al*., 1997a, 1997b), while classification systems developed by HELCOM for the Baltic Sea, by the Barcelona Convention for the Mediterranean Sea, and by OSPAR for the Northeast Atlantic are also included (see Figure 17.1).

Figure 17.1. Diagram showing the relationship between several classification systems.



Within EUNIS, marine habitats are distinguished at different levels. At the upper level, marine habitats are distinguished from the groups of terrestrial habitats (in total nine). At the second level, the water column is distinguished from the seabed, while the key criteria for further division are depth and substrate. This results in seven main categories of marine habitats at level 2 (Figure 17.2). In level 3, physical conditions, such as salinity and exposure, are introduced. An example of level 3 is given in Figure 17.3. From level 4 on down, biological characteristics of the habitat start to play a role.

Table 17.1 gives a typical example of the levels 3–7 currently distinguished within EUNIS. At present, the classification is still incomplete and partly under development. EEA is aware of the need for validation of the proposed classification. To this purpose, the EUNIS classification has been published on the World Wide Web for comments and feedback. EEA has decided to keep the current classification down to level 3 fixed for a review period of twelve months. During this period there is opportunity for proposing new units at the levels 4 and 5; these should be slotted into the existing framework. The new units have to meet the general criteria used in the current classification, and duplicates of the standing classification are not allowed. In September 2000, a joint OSPAR/ICES Workshop on Habitat Classification will

Figure 17.2. EUNIS Habitat Classification: criteria for marine habitats to Level 2, resulting in seven main categories.

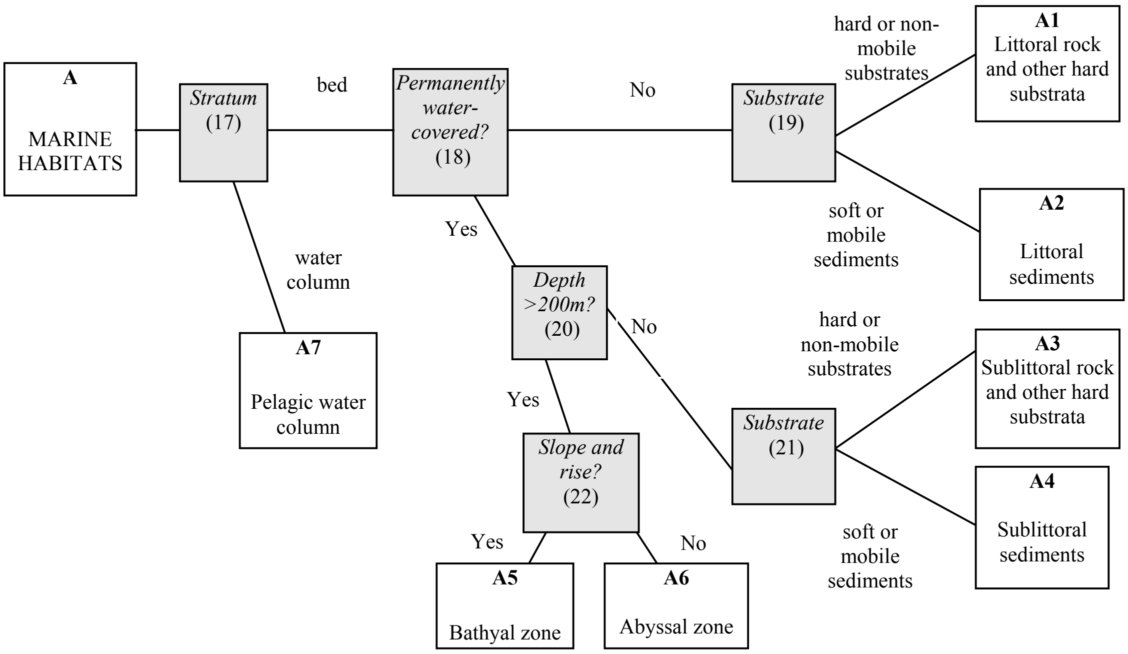
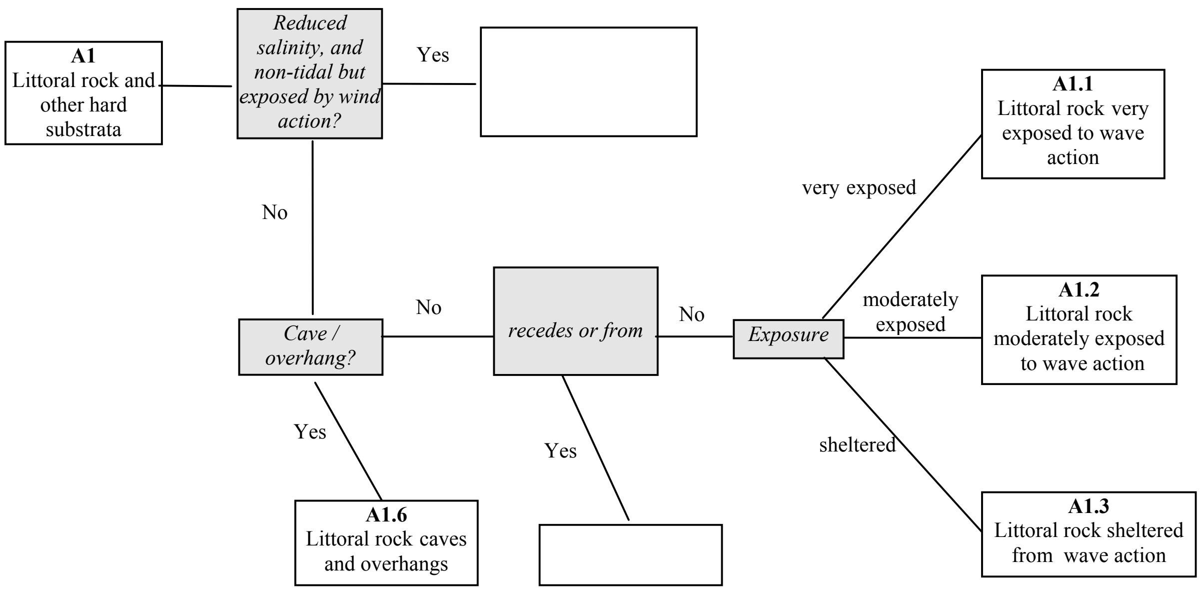


Figure 17.3. EUNIS Habitat Classification: criteria for littoral rock and other hard substrata (A1) to Level 3.



aim at further development of the classification. As to the matter of the validation of the EUNIS level 3 classification, a field testing programme will be run by EEA, starting by the end of April 2000. Furthermore, mapping of the OSPAR area at the level 3 will be an important test for the consistency of the classification.

Table 17.1. Example of the EUNIS classification system from level 3 on down to maximum level 7.

level 3	
A1.2	Littoral rock moderately exposed to wave action
level 4	
A1.2/B-MLR.BF	Fucoids and barnacles on moderately exposed littoral rock
level 5	
A1.2/B-MLR.BF.Fser	[*Fucus serratus*] on moderately exposed lower eulittoral rock
level 6	
A1.2/B-MLR.BF.Fser.Fser	Dense [*Fucus serratus*] on moderately exposed to very sheltered lower eulittoral rock
level 7	
A1.2/B-MLR.BF.Fser.Fser.Bo	[*Fucus serratus*] and under-boulder fauna on lower eulittoral boulders

(2) ARC classification under development

Sponsored by the Ecological Society of America and the NOAA Offices of Habitat Conservation and Protected Resources, discussions have started in the United States during the Aquatic Restoration and Conservation (ARC) Workshop to develop a framework for a national marine and estuarine ecosystem classification system to be used for monitoring habitats, in order to help managers to protect and conserve threatened ecosystems. As the EUNIS classification does not provide for a number of major habitat complexes in the USA (e.g., coral reefs, mangroves), it was decided to explore the feasibility of a classification system better adapted to North American conditions. Starting from the consensus that a classification system would provide a useful common language for describing habitats and a framework for interpreting ecological function, ARC developed a prototype of a marine and estuarine habitat classification system (Allee, 2000). This prototype is still very much under discussion. During 2000, a second ARC Workshop will give further consideration to the classification under development. The ARC system has the following principles:

1) the system is a blend of theoretical and pragmatic, as well as physical and biotic structuring variables;
2) it distinguishes up to thirteen levels (Table 17.2);
3) the twelfth level considers substratum and ecotypes;
4) the thirteenth level considers modifiers and eco-units:
    a) possible modifiers may be temperature, local energy regimes, salinity, history of extreme events, etc.,
    b) an eco-unit is the smallest element of the ecosystem as a whole. It represents the biological community that is the product of the physical and biotic variables above it, and is the closest approximation of the ultimate conservation target;
5) the classification system is structured to allow aggregation at different levels depending on the amount of data available on an ecosystem;
6) aggregating at higher levels results in more general information. However, as more specific information becomes available, more specific categorization can occur. This was necessary because the amount of information available on many ecosystems is limited. To accommodate this practical need, the position in the hierarchy of some of the variables is somewhat arbitrary and is based on the probability of the information being available.

Comments by ACME

The classification systems EUNIS and ARC have been discussed by the Study Group on Marine Habitat Mapping (SGMHM). Major comments with regard to EUNIS, as accepted by ACME, are:

- The EUNIS classification provides a common ground for the development of an ICES habitat classification;
- Its applicability, however, is as yet fairly limited, since EUNIS is primarily a European-based classification system with only the temperate zone;
- For large areas, data still appear to be missing and important characteristics of ecosystems are not dealt with properly in EUNIS (e.g., the dynamics and different features of anoxic sediments in the Baltic Sea);
- It needs to be assessed whether the EUNIS classification is going to work out well for the Northwest Atlantic (USA and Canadian) waters;
- EUNIS classification at level 3 could be taken as a template for the development of a classification system that can be expanded to the entire ICES area.

Table 17.2. Proposed marine and estuarine ecosystem classification system (from Allee, 2000).

1. Life Zone	1a. Temperate 1b. Tropical 1c. Polar
2. Water/ Land	2a. Terrestrial 2b. Water
3. Marine/ Freshwater	3a. Marine/Estuarine 3b. Freshwater
4. Continental/ Non-Continental	4a. Continental 4b. Non-Continental
5. Bottom/ Water Column	5a. Bottom (Benthic) 5b. Water Column
6. Shelf, Slope, Abyssal	6a. Shallow – on or over the continental shelf; < 200 m 6b. Medium – on or over the continental slope; 200–1000m 6c. Deep – on or over the rise and deeper features; > 1000 m
7. Regional Wave/ Wind Energy	7a. Exposed/Open – open to full oceanic wave or wind energies 7b. Protected/Bounded – protected from full wave or wind energies
8. Hydrogeomorphic/ Earthform Features	8a. Continental – Nearshore (surf zone); Inshore (rest of shelf); Straight or partially enclosed shorelines; Lagoons; Fjords; Embayments; Estuaries – Shore zone; Offshore zone; Delta; Carbonate settings; Outer continental shelf; Upper continental slope; Upper submarine canyon 8b. Non-Continental – Island (Volcanic; Low); Atoll; Submerged reef types
9. Hydrodynamic Features	9a. Supratidal – above high tides 9b. Intertidal – extreme high to extreme low water 9c. Subtidal – below extreme low water 9d. Circulation features – e.g., eddies
10. Photic/ Aphotic	10a. Photic 10b. Aphotic
11. Geomorphic Types or Topography	Cliff; Bench; Flat; Reef flat; Spur-and-Groove; Sand bar; Crevice; Slump; Rockfall; Terrace; Ledge; Overhang; Steeply sloping; Riverine; Fringe; Inland; Beach face; Dunes
12. Substratum and Eco-type	12a. Substratum (Not limited to this list) – Cobble; Pebble; Sand; Silt; Mud; Bedrock; Peat; Carbonate; Boulder; Biogenic; Organic; Anthropogenic 12b. Eco-type (Not limited to this list) – Coastal; Soft bottom; Hard bottom; Water column; Beach; Mangrove; Wetland; Seagrass bed; Coral reef; Kelp bed; Mud flat
13. Local Modifiers and Eco-unit	13a. Modifiers (Not limited to this list) – Temperature; Local energy regimes – waves, tides, current; Salinity; Nutrients; Alkalinity; Roughness/relief; Dynamism; Edge effects – from adjacent areas; Anthropogenic disturbances; Biological interactions; Extreme events – history 13b. Eco-units – Unlimited representation of species resulting from modifiers applied at the above hierarchical levels.

Comments with regard to the ARC system are:

- The ARC classification system is still under development and will be modified taking into account the remarks of SGMHM, as adopted by ACME;
- The differences with EUNIS are not as large as they appear to be at first glance; while ARC takes a different approach to ranking habitats, the many levels distinguished by ARC are comparable to parameters in the EUNIS system;
- A major difference is that EUNIS uses the criteria exposed/non-exposed and sediment type right in the beginning, while the ARC Workshop was in favour of using these criteria lower down in the decision tree;
- ARC includes all life zones (temperate, tropical, and polar zones);
- ARC is not purely hierarchical;
- The concepts of ecotypes, modifiers, and eco-units are promising. However, the term “ecotype” is confusing and has to be changed. Ecotype is an existing word with a different meaning (Baretta-Bekker *et al*., 1998).

Developments in Habitat Mapping

Several habitat mapping projects within the ICES area are running or planned, for example, the projects in Belgium, Canada, the Netherlands, Norway, and the UK, as described below. Most of them are small scale mapping projects using different classification systems. This is a summary of the more detailed information contained in the SGMHM report.

Belgium. The project HABITAT evaluates the evolution of a protected benthic habitat. The protected area in question is the Western Coastal Banks in the Belgian coastal zone. The area is an important overwintering place for the common scoter (*Melanitta nigra*). In order to describe the T0 situation of the benthic habitat, data on bathymetry, sedimentology, and hydrodynamics are gathered together to make a bathymetry map and a geomorphological map. Correlations between digital sidescan sonar recordings and sedimentology and between sedimentology and macrobenthos are established. The project HABITAT will examine which information about macrobenthos can be gathered from sidescan sonar recordings and try to develop a standardized macrobenthos sidescan sonar interpretation and create a generalized “habitat” map of the complete protected area.

Canada. SEAMAN is a multiyear, multidisciplinary research proposal, developed by three government departments (Fisheries and Oceans, Natural Resources Canada, and the Department of National Defence). SEAMAN will provide basic seabed mapping data for sustainable management of offshore resources. Applications include fisheries management, offshore mineral resource assessment, selection and management of Marine Protected Areas, siting of offshore structures (platforms, pipelines, and cables) and national defence. Systematic data acquisition will incorporate multibeam bathymetry, sidescan sonar, high-resolution seismic reflection profiling, grab sampling, and bottom photography of characteristic sediments and biotopes. Data and interpretative maps will be archived in GIS format and output as electronic charts, maps, and interpretative reports. Maps for bathymetry, habitats, contaminants, sediment types, mineralogical resources, and geological features have many uses, e.g., increasing the efficiency of scallop fisheries.

The Netherlands. Within the framework of the Trilateral Cooperation on the Protection of the Wadden Sea (Denmark, Germany, the Netherlands), a Trilateral Monitoring and Assessment Programme (TMAP) has been developed. The TMAP comprises a wide set of specific parameters providing information on the chemical and biological status and developments of the Wadden Sea. In addition, human activities are monitored. Presently under development is a set of so-called general parameters. These comprise, among other properties, aspects of the geomorphology (extent of high/low tidal flats, extent of sediment types) and hydrology (salt marsh, inundation frequency, wave climate) of the area, climate, and weather conditions.

HABIMAP is a GIS-guided method to prepare ecotope and habitat maps. There are at present maps for the North Sea, the Wadden Sea, and the Western Scheldt (de Jong, 1999). HABIMAP has its own classification system which can easily be converted up to level 3 of the EUNIS classification system (Figure 17.4). For level 4, assemblages are used derived from Holtmann *et al.* (1996) (Figure 17.5). This application could be used as a general framework by providing a common basis for mapping at the proposed EUNIS classification level 3, at the same time enabling the user to incorporate regional maps with more detailed levels of classification.

Norway. Mapping of deep-water coral reefs is being carried out. The presence/absence of coral reefs along the Norwegian continental shelf has been investigated. To keep the costs down, information on the presence of *Lophelia pertusa* has been collected from fishermen, literature and reports from Statoil, the Norwegian Directorate of Fisheries, and the Institute of Marine Research. Reports of reefs damaged by bottom trawling have been mapped as well. The large reef complex on the Sula ridge has been mapped by means of multibeam echosounding. This rapid mapping combined with video recordings of these areas has led directly to the protection of two reef areas.

The marine habitat mapping project MAREANO is planned by five major governmental institutions and will provide information on bathymetry, marine habitats, biological diversity and natural resources coupled to the seabed, baseline data on contaminants, baseline data on sediments, sediment types, mineralogical resources, and geological features. The study area is on the mid-Norwegian shelf and the Vøring Plateau.

United Kingdom. At CEFAS a three-year research project is under way. It is to characterize the seabed using various physical and geophysical techniques. Based on the results of different techniques (sidescan sonar and the seabed discrimination systems, RoxAnn and QCT View), the seabed of several areas is divided into acoustically distinct regions. The acoustic outputs from each region are "translated" into sediment types from cobbles and gravels to sand and mud. Biological samples were taken in each of the acoustically distinct regions.

In Northern Ireland, habitat mapping is being undertaken in the inshore waters and sea loughs. The two principal requirements are the need to map shellfish resources (e.g., *Nephrops*) and special areas of conservation (SACs).

A project on environmental risk assessment in the offshore oil and gas industry in the North Sea is under way. The project is using a GIS to visually represent existing species, and physical data, categorizing these areas into biotopes using the suggested systems and comparing them to statistically analysed groupings. The biotopes are then assessed for sensitivity to physical, chemical, and biological factors from the same system as the MarLIN project (Hiscock *et al.*, 1999). The final representation is designed to be accessible to several levels of users to aid its application.

Discussion

The classification of marine habitats on a large-scale area is feasible if it is based primarily on hydrographic and geological features of the system under consideration. A classification system using these features will generally fit into the current EUNIS classification at levels 2 and 3, and will provide a solid basis for habitat classification within the ICES area. Physical habitat description together with biological ground-truth sampling could form the basis for the creation of large area predicted-biotope maps. Biological data should then be added, overlaying the physical habitats, at the level of functional groups (level 4) or species (level 5). This provides a large challenge to the scientists involved as it is recognized that, especially in the deeper parts of the Atlantic, the availability of biological data is very poor. The ACME feels that by approaching the classification issue in this way, working from coarse to fine, the development of a classification system that will be useful to ICES can go forward.

Classification and mapping are two concepts that are closely interrelated: the size of the basic mapping unit determines how detailed the classification will be. On the one hand, a large basic mapping unit will result in a coarse, large-scale map without details. For instance, a large-scale map can usually do without much underlying ("biological") data, thus resulting in a physical habitat-type of map (Figure 17.6). On the other hand, biological ground-truthing means collecting field data in areas restricted in size, with smaller-scale maps as a result (Figures 17.4 and 17.5). A flexible classification system (i.e., a multilevel classification) should enable end-users to produce both detailed, small-scale biological maps as well as large-scale physical habitat maps. A hierarchical classification system will allow this, because it has the ability to nest smaller-scale maps (e.g., Figures 17.4 and 17.5) within a larger-scale (e.g., Figure 17.6) map. In a hierarchical classification system, a particular level of classification will include a range of lower levels.

Conclusions

Based on the material above, the ACME accepted the following conclusions of SGMHM:

1) EUNIS classification at level 3 is suitable for use as a template for the development of a classification system to cover the entire ICES area.
2) There is flexibility for the inclusion of new units at levels 4 and 5 in the existing framework.
3) Elements of ARC can be used for the definition of such new units.
4) The classification system to be developed must both form a solid basis for primary classification of large-scale areas and allow a sufficient amount of detail to be of use in restricted areas. A hierarchical system with nested maps can fulfill this requirement.
5) Mapping activities should be carried out at scales from coarse to fine:
    a) start with the production of large-scale (predicted) biotope maps based on the physical characteristics of the area, in combination with biological ground-truth sampling,
    b) refine the large-scale biotope maps to produce small-scale maps by overlaying them with biological field data,
        i) detailed biotope mapping applications can be done with swath systems, such as sidescan sonar, multiple narrow-beam swath bathymetry, and seismic sidescan sonar; their costs are high and post-processing is time-consuming and needs experience for interpretation; other techniques are remote video recording, grab sampling, etc.
6) In collecting (biological) data, attention should primarily focus on the shelf seas and the slope, as these are the marine areas that experience the highest pressure from human activities.

Recommendations

The ACME recommended that high priority be given to the initiation of the development of a marine habitat classification system for the ICES area, taking the EUNIS classification as a template, and to further build on the classification at levels 4 and 5. This should also include continued participation in the evaluation of EUNIS level 3, by mapping and testing. The Benthos Ecology Working Group (BEWG) and the Working Group on the Effects of Extraction of Marine Sediments on the Marine Ecosystem (WGEXT) should contribute to the verification of the EUNIS classification.

Efforts to develop marine habitat mapping should cover the entire ICES area and should be based on the elaboration of a generally accepted classification system.

Additional comments

The ACME noted with great appreciation the vast interest within the ICES scientific community, and the significant progress, in habitat classification and habitat mapping in the ICES area. Development of this issue is crucial for further progress in understanding the effects of exploitation of living and non-living marine resources, effects of large-scale technical constructions, improvement of Environmental Impact Assessments, as well as the establishment of Marine Protected Areas.

The ACME also noted with appreciation the good cooperation between SGMHM and WGEXT and the development of WGEXT activity on seabed characterization and biotope mapping.

The ACME is aware of recent developments on habitat/biotope mapping (EU CORINE, The SEA-SEARCH Habitat Guide (Earll, 1992), EEA EUNIS Habitat Classification, HELCOM Red List of Marine and Coastal Biotopes and Biotope Complexes of the Baltic Sea, Belt Sea and Kattegat). The ACME considers such information to be necessary for decision-makers regarding sustainable development policies related to the exploitation of the marine environment.

The ACME noted with appreciation the preliminary announcement about a habitat mapping workshop to be organized by the Institute of Marine Research, Bergen, Norway.

The ACME recommends an interdisciplinary effort including benthic ecologists, geologists, fishery scientists, and nature conservation experts in the further development of marine habitat mapping.

Figure 17.4. Habitat map of the southern North Sea; EUNIS classification at level 3.

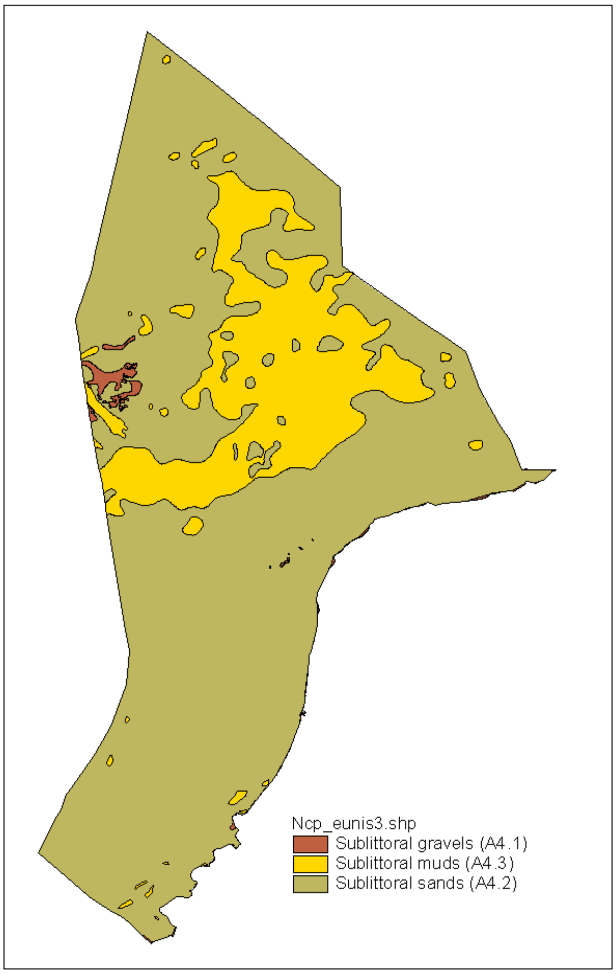


Figure 17.5. Habitat map of the southern North Sea; classification at level 4, using assemblages according to Holtmann *et al*. (1996).

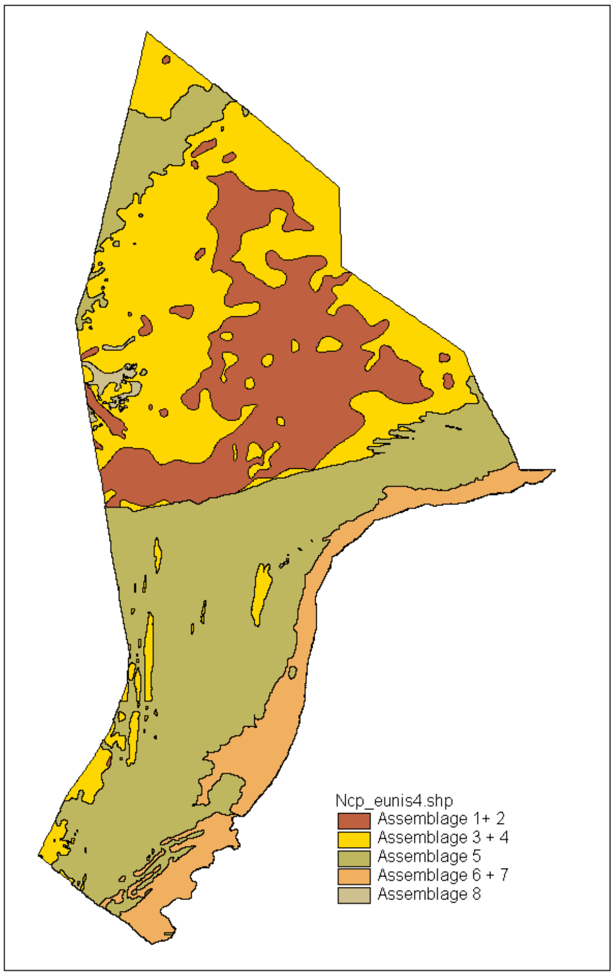
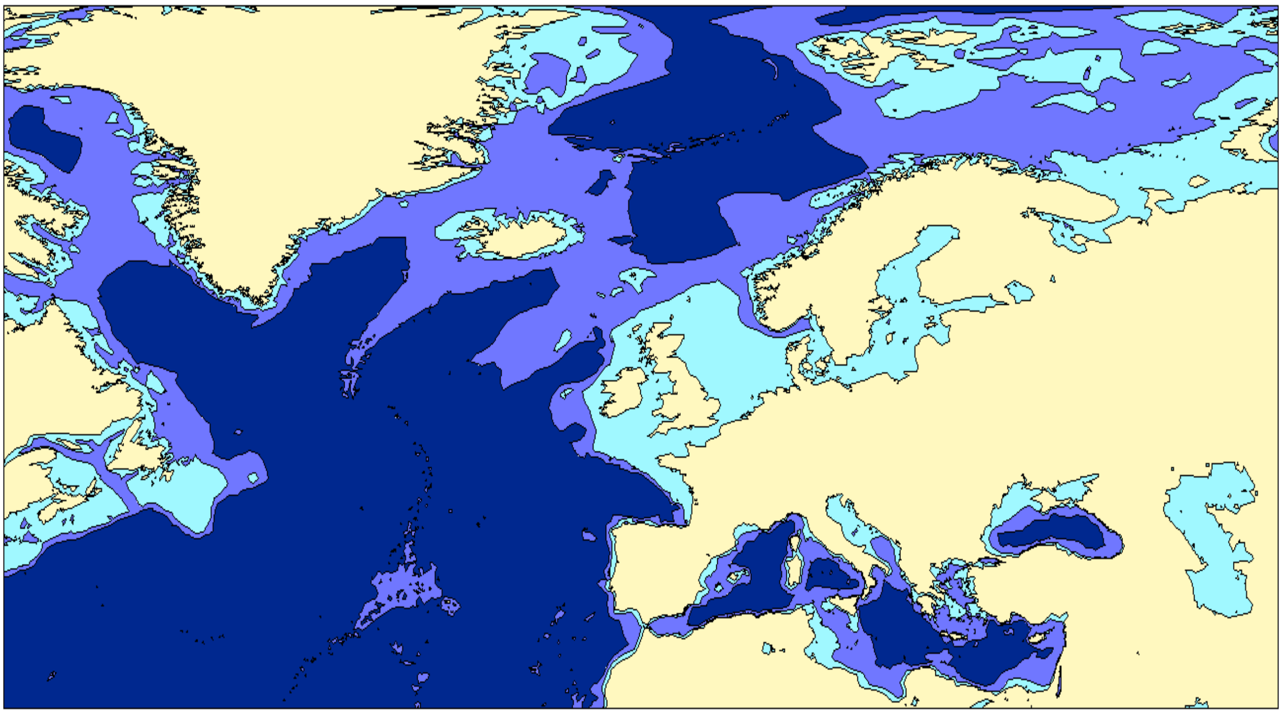


Figure 17.6. Map of the ICES area, with depth zones < 200 m, between 200 m and 2000 m, and > 2000 m.



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18 ENVIRONMENTAL AND ECOSYSTEM ASSESSMENT

18.1 Ecosystem Effects of Fishing in the Baltic Sea

Request

Item 3 of the 2000 requests from the Helsinki Commission: to prepare Chapter 9 on “Marine fish migratory and freshwater species in the Baltic Sea area” for the Fourth Periodic Assessment of the State of the Baltic Marine Environment, 1994–1998, including ecosystem effects of fishing activities.

Source of the information presented

The 1999 report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO) and ACME deliberations.

Status/background information

Environmental effects of the Baltic Sea fisheries are not known with certainty, but are considered to be substantial, particularly 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 and weak recruitment. Changes in the cod stock have undoubtedly influenced its primary fish prey—sprat and herring. However, these species themselves are targets of directed fisheries, so it is not straightforward to differentiate in their population dynamics the direct effects of their fisheries from responses to the reduced abundance of cod. Sprat and herring are dominant zooplanktivores in the ecosystem, and zooplankton populations are expected to be affected by the changes in abundance of their predators. Again, however, it is not easy to partition the effects of changing zooplanktivore abundance from the effects on zooplankton of changing nutrient availability and hydrographic conditions. Other effects of fishing, on invertebrates, marine mammals, and birds, are poorly known. Multispecies assessment models have been used to evaluate predator-prey interactions among cod, herring and sprat, and how these interactions may be affected by fisheries. However, these models do not include lower trophic levels, so they cannot provide insight into some types of possible effects of fishing on the entire food web.

Many of these difficulties characterize all exploited marine ecosystems, and have been explored in depth by ICES in evaluating effects of fishing on the North Sea ecosystem. However, differences between the North Sea and the Baltic Sea in their hydrographic regimes and the structure and diversity of their biological communities, mean that it was not considered appropriate to transfer uncritically conclusions from work in the North Sea to the Baltic situation.

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 nets and longlines while feeding in the sea and with coastal gillnets and traps during the spawning run.

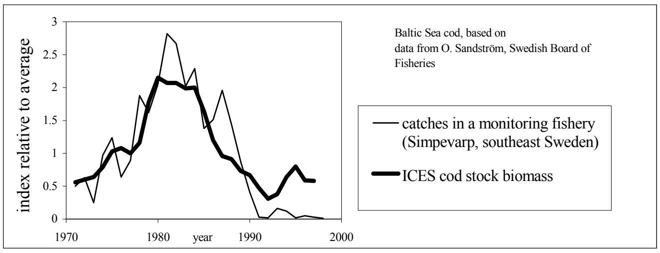
The coastal fishery targets a variety of species with a mixture of gears including fixed gears (e.g., gillnets, pound and trap nets, and weirs) and Danish seines. The main species exploited are herring, flounder, turbot, cod, eel, pike, perch, pikeperch, smelt, and whitefish. In addition, there are also 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.

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. Three potential consequences of this are reductions in offspring production, changes in area occupied, and changes in food web dynamics. In the Baltic, there are strong indications that the fishery has caused all three types of changes (Hansson, in press). With regard to food web effects, when cod, the dominant piscivorous fish in the Baltic, had low reproduction success owing to hydrographic conditions (Sparholt, 1996; Jarre-Teichman *et al*., 1997), 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-controlled 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 with 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).

With regard to effects on distribution, a consequence of the decreased cod stock is that the species has almost disappeared from some of its normal distribution areas, i.e., the Archipelago Sea, the Bothnian Sea, and coastal areas of the Baltic Proper (O. Sandström, pers. comm.); see Figure 18.1.1. For effects on the production of progeny, ICES has identified B_{lim} values for western Baltic cod and sprat stocks, designating spawning biomasses below which there is evidence that recruitment is impaired (ICES, 1999b, 1999c). At present, the biomass of the eastern Baltic cod stock is below B_{lim}, and was for a period in the early 1990s as well. Excessive exploitation contributed greatly to the depressed condition of the stock in both cases. As noted above, however, environmental conditions were also unfavourable for cod reproduction, and the effect of fishing on reduced offspring production has not been disentangled unambiguously from environmental influences on recruitment. Sprat was below the value now used as B_{lim} in the early 1980s, and again excessive fishing combined with poor recruitment to move the stock to a condition where the production of recruits was impaired.

Figure 18.1.1. 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.

Effects due to by-catch and discarding

The total by-catch of fish is unknown for the Baltic Sea fisheries. Earlier reports generally contained few data on this topic, so quantitative estimates could not be derived. However, the EU has supported several very recent studies of by-catch, most with results not yet published. The Study Group on Estimation of the Annual Amount of Discards and Fish Offal in the Baltic Sea (SGDIB) has compiled much of the information from these studies. Their results are summarized in Section 11 of this report. The results in Section 11 are primarily from the major fisheries for cod, herring, and sprat. There are, however, some smaller and coastal fisheries that have very high proportions of by-catch. In the fishery for vendace roe in the Bothnian Bay, for example, the by-catch is 92 % by weight (O. Sandström, pers. comm.).

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 the oxygen concentrations in bottom waters. As the analysis reported in Section 11 shows, this is a negligible problem.

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. ICES has recently conducted a comprehensive review and provided advice on these effects in the North Sea (see Section 5, above). By comparison, effects of bottom gears in the Baltic are expected to be smaller than in the North Sea, because beam trawling does not occur and the benthos is generally dominated by smaller organisms than in the North Sea.

Towed gears that penetrate the sediment considerably impact both benthic infauna and epifauna, whereas other types of gears (e.g., gillnets and trap nets, Danish seines) 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 Kiel Bight. The applicability of these findings to the larger Baltic Sea requires additional studies in other parts of the Baltic Sea.

- There was evidence of sediment disturbance in the Kiel Bight, including remobilization of nutrients, and increased release of nutrients and organic material was followed by an increase in oxygen consumption (Krost, 1990).
- Biological responses of the invertebrate community to trawling activity in the Kiel Bight were also documented. 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 damage. An increase in predatory and scavenging species feeding on dying and dislocated fauna was also documented in this area (Rumohr and Krost, 1991).

Effects on seabirds

Fishing nets, in particular set nets, cause considerable mortality of seabirds in the Baltic Sea. Oldén *et al*. (1988) estimated that about 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 type of fishery 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 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).

Effects on marine mammals

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 and the by-catches are possibly underestimated; thus, these by-catches may be unsustainable (ICES, 1997).

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, 1995). 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 number of seals killed in the Swedish coastal fishery is 400 animals (O. Sandström, 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, they do not appear to constitute a major threat to the seal populations, because seal numbers are increasing (Helander and Härkönen, 1997).

Seals are also known to take salmon and other fish from gillnets and traps (ICES, 1999b, 1999c), so fishing can also be considered to affect seals through the provision of food.

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. Sustainable fisheries must be managed within the boundary conditions set by low species diversity and little redundancy, and strong environmental perturbations whose occurrences can be predicted poorly, or not at all. The fisheries viewed together have not taken adequate acknowledgement of these boundary conditions, and have contributed substantially to large ecosystem changes. The primary impact of the fishery has thus been the direct removal of target species, which has occurred at excessive rates several times. Secondary effects are impacts of bottom trawling and by-catches, in particular of harbour porpoises.

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 fourteen 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. This management must be built on an adequate scientific basis, which can only be provided by well-integrated research and monitoring programmes. These programmes should be designed and conducted in a full ecosystem framework, with full consideration of the information needed to implement an ecosystem approach to management. Because the Baltic Sea is one of the most intensively studied areas of the world's oceans, experiences from this broader and integrated approach to research, monitoring, and management will probably be of interest elsewhere.

Recommendations

The key recommendation 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. 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. Such work should continue and be given priority.

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18.2 Chapter on Fish in the Baltic Sea for the HELCOM Fourth Periodic Assessment

Request

Item 3 of the 2000 requests from the Helsinki Commission: to prepare Chapter 9 on “Marine fish migratory and freshwater species in the Baltic Sea Area” for the Fourth Periodic Assessment of the State of the Baltic Marine Environment, 1994–1998, comprising:

9.1 Exploited species

9.2 Non-commercial species

9.3 Diseases and parasites

9.4 Effects of rearing upon wild populations

9.5 Ecosystem effects of fishing activities.

The draft chapter should be transmitted initially to HELCOM according to the timetable for the Fourth Periodic Assessment.

Source of the information presented

A draft chapter on “Marine fish migratory and freshwater species in the Baltic Sea Area”, prepared on the basis of material provided by the Baltic Fisheries Assessment Working Group (WGBFAS), the Baltic Salmon and Trout Assessment Working Group (WGBAST), the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO), the Working Group on the Application of Genetics in Fisheries and Mariculture (WGAGFM), the Working Group on Environmental Interactions of Mariculture (WGEIM), and the Working Group on Ecosystem Effects of Fishing Activities (WGECO), and ACME deliberations.

Status/background information

The ACME reviewed the compilation of material prepared for this draft chapter by the above-mentioned Working Groups. This compilation provided an extensive amount of material, and the chapter was considerably longer than the number of pages allocated by the HELCOM Project Group for the Fourth Periodic Assessment for this chapter. Accordingly, the ACME decided that only a brief summary of much of the material should be included in the chapter for the Fourth Periodic Assessment, in order to adhere to the page limit. However, as the detailed material was considered to be informative and useful, the ACME agreed to include portions of this material in its report. Accordingly, the following information has been included in various places in this report, as follows:

1) a report on fish diseases in the Baltic Sea (Annex 6, with a brief summary in Section 9.4);
2) information on the effects of mariculture activities in the Baltic Sea (Section 14.2);
3) a summary of information on the genetic effects of the release of cultured fish in the Baltic Sea (Section 15.1);
4) a summary of the current understanding of ecosystem effects of fishing activities in the Baltic Sea (Section 18.1).

The ACME then adopted a considerably shorter draft chapter for transmission to HELCOM for use in the Fourth Periodic Assessment. However, as this chapter was still too long, the ACME requested the ICES Secretariat to further reduce the length of the chapter, in consultation with the HELCOM Project Group for the Fourth Periodic Assessment.

Need for further research or additional data

The ACME noted that some information had been supplied by WGEIM concerning concentrations of dioxins in Baltic herring, however, the source of this information was not apparent. Given the reported values, the ACME agreed that further, quality controlled data should be compiled to evaluate the levels of these contaminants in Baltic fish.

18.3 ICES Environmental Status Report

Request

During the past few years, several ICES Working Groups have agreed to contribute to an ICES Environmental Status Report, which will be updated annually or more frequently, depending on the subject matter. The Environmental Status Report is published on the ICES website as material becomes available.

18.3.1 Oceanographic conditions

Source of the information presented

The ICES Annual Ocean Climate Status Summary for 1999/2000, available on the ICES website at http://www.ices.dk/status, and ACME deliberations.

Status/background information

Summary

The North Atlantic marine climate is largely controlled by the so-called North Atlantic Oscillation (NAO), which is driven by the Azores High and the Iceland Low pressure cells. In a given year, the intensity of the NAO is simply described by the pressure difference between these two main cells and this NAO index is normally measured between Lisbon (Portugal) and Stykkisholmur (Iceland). Over the past four decades, the NAO index has shown a progressive increase from its most extreme and persistent negative state in the 1960s to its most extreme positive state during the late 1980s to early 1990s, allowing scientists to construct a picture of the kinds of ocean responses to expect under either phase of the NAO.

Following its long period of amplification, the NAO index suddenly underwent a sharp decrease to a short-lived minimum in the winter of 1995/1996 with radical and recognizable changes in the North Atlantic. Since that temporary minimum, a steady recovery towards positive values of the NAO and the resumption of many of the ocean climate features associated with that positive phase have been observed.

However, the recovery is still not complete. Whereas warm, saline conditions have resumed along the eastern boundary of the North Atlantic to the Barents Sea as expected, conditions in the Northwest Atlantic are still abnormal for the positive state of the NAO. Instead of a cold and strong northwesterly airflow promoting intense cooling in winter as it did in the early 1990s, northwesterlies are now mainly confined to the east of Greenland, such that the Labrador Sea is occupied by light or southerly anomaly winds. It is within this overall context that the climatic status of individual sea areas in 1999 should be viewed.

The main features for 1999/2000 were:

- West Greenland air and sea temperatures were still very mild compared with average conditions;
- in the Northwest Atlantic, air and sea temperatures have been exceptionally warm, with a reduced occurrence of sea ice;
- in the Labrador Sea, open ocean convection was weak and shallow; near-surface waters were warmer and fresher than in 1998;
- in the Atlantic waters both south and north of Iceland, temperatures and salinities were above average and among the highest observed in recent decades;
- in the Bay of Biscay, air temperatures were warmer than average and surface waters were the warmest observed from 1992–1999;
- in the Rockall Trough (Northeast Atlantic), the temperatures were the highest recorded in the region since measurements began 25 years ago and salinities remain considerably above average;
- further north, Atlantic water at the northwest European shelf edge was also warm and saline;
- in the North Sea, 1999 was characterized by warmer than average surface temperatures throughout the year, particularly in September;
- in the Norwegian Sea, temperatures continued to be above the long-term average, while salinities continued to decline;
- temperatures in the Barents Sea quickly rose at the beginning of 1999, starting in the west, and warm conditions generally prevailed throughout the area all year;
- in the Greenland Sea and Fram Strait, the temperature and salinity of the northward flowing Atlantic water increased, whereas the southward return flow from the Arctic continued to cool and freshen.

18.3.2 Harmful algal blooms

Source of the information presented

The 2000 report of the ICES/IOC Working Group on Harmful Algal Bloom Dynamics (WGHABD) and ACME deliberations.

Status/background information

The ACME noted that it has published updated decadal maps of harmful algal events in the ICES area in its report for the past three years. The most recent updates of these maps are available on the ICES website (http://www.ices.dk/status/decadal/) as a part of the

Environmental Status Report and, thus, the ACME will no longer include them in its own report.

18.3.3 Fish disease prevalence

Source of the information presented

The 2000 report of the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) and ACME deliberations.

Status/background information

At its 2000 meeting, WGPDMO considered the development of proposals for the inclusion of maps of the distribution of fish and shellfish diseases of concern for mariculture and temporal trends of wild fish diseases of concern for marine environmental monitoring programmes for inclusion in the ICES Environmental Status Report and on the ICES website.

A series of maps were produced illustrating temporal trends of the diseases lymphocystis, epidermal hyperplasia/papilloma, and skin ulcerations in dab (*Limanda limanda*) from the North Sea and the English Channel, the spatial distribution of the oyster diseases *Bonamia oestrea*, *Martelia refringens,* and *Perkinsus marinum* covering all ICES countries, and the spatial distribution of marine Viral Haemorrhagic Septicaemia (VHS) covering all ICES countries.

Diseases of dab were selected because they are used in national marine environmental monitoring programmes carried out by ICES Member Countries bordering the North Sea and have been recommended for inclusion in the OSPAR Joint Assessment and Monitoring Programme (JAMP). Data used were extracted from the ICES Environmental Data Centre.

The diseases of shellfish were selected as they are some of the most important diseases in oyster culture and marine VHS was selected as it may be an emerging disease problem. Data for these maps have been provided by WGPDMO members. It is intended that the maps will be expanded to include other important shellfish diseases and diseases in mariculture.

After review, the ACME accepted these maps of the distribution or temporal trends of several diseases of fish and shellfish for inclusion in the Environmental Status Report. These maps are attached as Annex 8.

Need for further research or additional data

It is the intention to expand the maps to include other diseases in marine fish farming. However, the registration and reporting of these diseases are not uniform in the different countries. This problem has to be dealt with in the near future to ensure that the illustrations will be comparable from country to country.

18.4 Structure, Process, and Limitations of Environmental Assessments and Production of Environmental Quality Status Reports

Request

ICES has been closely involved in the peer review of various marine environmental quality status reports and therefore is in a good position to advise international monitoring organizations on ways in which the structure and process of Quality Status Report (QSR) production can be improved.

Source of the information presented

Various reports, as listed under “Documents considered”, and ACME deliberations.

Status/background information

The ACME reviewed the methods whereby marine environmental assessments have been produced in the ICES area. It is clear that such assessments have not proceeded smoothly in the past, and it is possible to envisage a number of improvements which could usefully be implemented before further regional or sub-regional assessments are undertaken.

The substantive issues concerning environmental assessments fall into several categories: Objectives; Report Structure; Data Handling; Assessment Process; Limitations in our abilities. These issues are briefly summarized below and described in full in Annex 9.

a) Objectives

- Must be clear and transparent;
- Should not be changed during the assessment process;
- Should be achievable with available resources.

b) Report structure

- Arrangement by discipline is appropriate for initial assessments;
- Subsequent assessments are better organized thematically;
- Regular thematic assessments on different subjects are more useful than infrequent comprehensive reports.

c) Data handling

- All sampling, analysis, and reporting protocols should be agreed in advance, and considered mandatory;

- Data collected should be for the same period as the report;
- Underpinning socio-economic and other data that do not derive from marine monitoring programmes should be sourced at an early stage and formatted appropriately;
- Biological and chemical monitoring data should be collected from the field in an integrated fashion, not as separate data sets derived from different samples;
- All staff involved in data gathering and manipulation should use fully compatible spreadsheets, mapping software, and databases;
- Data transfer protocols (for e-mailing information) must be agreed in advance, and standardized;
- All sub-regions should use the same models for deciding priorities for further action.

d) Assessment process

- Draft assessments should be produced by small dedicated teams, if necessary under contract, using pre-arranged and mandatory assessment guidelines;
- If text, tables, or figures are, nevertheless, produced by many authors, they should all use standard software templates for the purpose;
- All draft report material should be placed, as soon as it is produced, on a website accessible to all relevant parties;
- All data should be placed on a freely accessible data management system with on-line data extraction protocols;
- Adequate resources and time must be provided for personnel conducting assessments;
- Due to turnover of personnel, it is much easier to produce annual update reports than infrequent full reports;
- ICES should continue to provide external peer review of assessment reports for quality assurance.

e) Limitations in our abilities

- Monitoring expertise and technology vary greatly between countries;
- Data available on trace organic inputs to the marine environment are only patchy;
- Many national monitoring data are not submitted to the ICES databases and are thus difficult to use in assessments;
- Maintenance of long-term monitoring data sets, which are vital for predicting climatic and anthropogenic impacts, is not always properly funded;
- Data useful for predicting the ecosystem effects of fisheries are sparsely available;
- Data available on the economic value of marine resources are very inadequate;
- The derivation and use of Background/Reference Concentrations, Ecotoxicological Assessment Criteria, and Ecological Quality Objectives require further refinement and thought before they can be considered useful in marine quality assessments;
- There is a scarcity of biological, and biological effects, data from outside the fisheries sector.

Documents considered

Boelens, R., and O'Sullivan, G. 2000. Quality Status Report 1999, Ireland. Experience gained—lessons learned. Marine Institute, Dublin, Ireland. 7 pp.

Enserink, L., Oudshoorn, B., and van der Valk, F. 1999. Regional Quality Status Report 1999—The current status of the Greater North Sea. ICES CM 1999/V:04. 5 pp.

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Reid, P.C., and Carlberg, S.R. 1999. The mechanics of QSR 2000. ICES CM 1999/V:06. 16 pp.

Salchow, R. 1999. Towards a first ocean-wide assessment. ICES CM 1999/V:01. 6 pp.

Recommendations

It is recommended that monitoring commissions and ICES Member Countries take note of the points raised above when planning and executing future marine environmental monitoring programmes. A detailed list of recommendations is given in Annex 9.

18.5 Review of Ecosystem Models as Basis for Choosing Metrics of Ecosystem Status and Evaluating Indirect Effects of Fishing

Request

This is part of continuing ICES work to understand ecosystem dynamics in relation to the effects of fisheries.

Source of the information presented

The 1999 report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO) and ACME deliberations.

Status/background information

Through WGECO, ICES has commenced a review of the principal models of ecosystem dynamics, to develop specific predictions based on each of them for the ecosystem effects of fishing. Some progress was made on this task, particularly with regard to developing a complex inventory and taxonomy of such models. However, much of the comparative testing and critical evaluation of all models remains to be completed.

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*. (2000a), shown in Figure 18.5.1. Additional material in the WGECO report assigns 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. Different families of models will address essentially different issues, or provide different insights into ecosystem operation. The WGECO report lists the principal properties of each class of model without reviewing their validity or usefulness. It also contains generic predictions about the ecosystem effects of fishing if each of the models were a correct description of the ecosystem. In cases where different families of ecosystem models produce predictions about similar ecosystem properties using very different types of information or assumptions, the key differences are identified but not evaluated against each other.

This inventory is valuable because of its completeness and systematic organization of a large field of research. However, it does not provide the basis for preferring some ecosystems models over others, except to the extent that one would choose a model which fits the data one happens to have. A case could be made that the ability of a model to make reliable predictions about the ecosystem properties of interest would be a better basis for selecting a model. However, making choices on such criteria would require a more thorough review of the models and their properties. Such a 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.

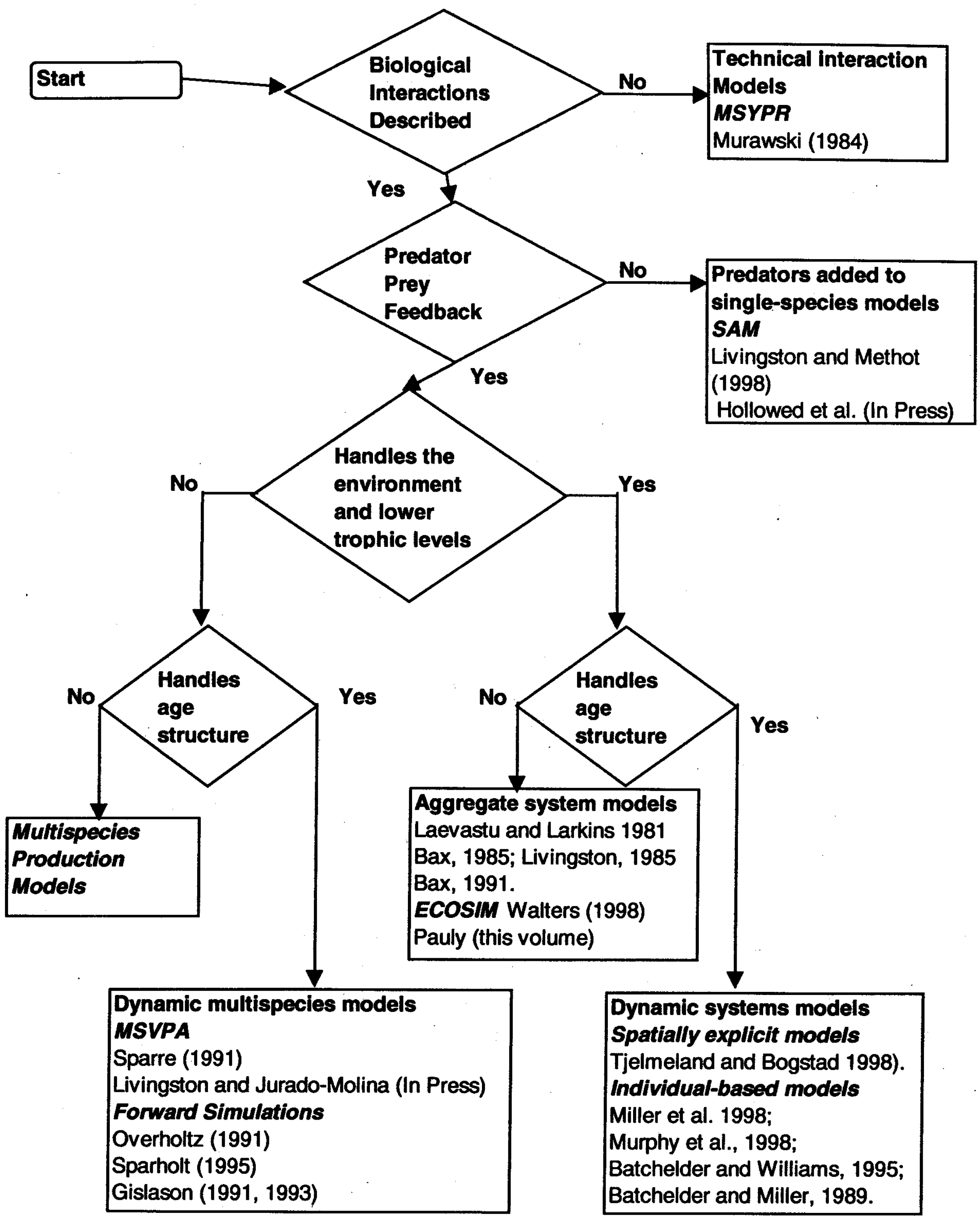
Only a few of the models mentioned in Figure 18.5.1 consider single species in a way that can be directly compared with current assessment models, dealing as they do in the most part with multispecies interactions. Multispecies models, therefore, provide a means of examining how fishing (or many other types of) disturbance might affect the emergent properties of ecosystems, in particular food-web dynamics and change in species diversity. This, in turn, might provide clear answers to questions about the risk posed to emergent ecosystem properties in systems where individual species are not at risk.

Further progress

A dedicated workshop seems to be a necessary forum to address 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. Before the workshop, preparations would have to be made to be able to 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.

The above work is necessary because ecosystem models are essential to tease out the role of environmental forcing, fisheries, and other anthropogenic perturbations on 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 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 entire 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.

Figure 18.5.1. 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 paper by Hollowed *et al.* (2000a).



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18.6 Methodology and Proposal for Ecological Quality Objectives for the North Sea

Request

The 2001 Work Programme from the OSPAR Commission contains two requests regarding ecological quality objectives.

Source of the information presented

The 2000 report of the Study Group on Ecosystem Assessment and Monitoring (SGEAM) and ACME deliberations.

Status/background information

Two Workshops, the Workshop on the Ecosystem Approach to the Management and Protection of the North Sea, held in Oslo in June 1998 (NMC, 1998), and the Workshop on Ecological Quality Objectives (EcoQOs) for the North Sea, held in Scheveningen in September 1999 (NMC, 1999), and SGEAM have contributed to the process of preparing Ecological Quality Objectives (EcoQOs) for the North Sea.

An ecological quality objective should reflect an ecosystem approach and integrate objectives for various ecosystem components, for example, within a multidimensional framework. It should result from a management decision based on scientific advice. The proposed definition is:

"The desired level of ecological quality relative to a reference level."

Within the OSPAR framework for EcoQOs for the North Sea, a set of ten issues have been identified:

1) reference points for commercial fish species;
2) threatened or declining species;
3) sea mammals;
4) seabirds;
5) fish communities;
6) benthic communities;
7) plankton communities;
8) habitats;
9) nutrient budget and production;
10) oxygen consumption.

These issues divide the ecosystem into manageable units, under which EcoQOs can be developed. The ACME is of the opinion that, in order to implement an ecosystem approach, all ten issues should be considered together in an integrated manner. Measuring individual EcoQOs would not necessarily comply with an ecosystem approach. It is also felt that further work is required to review the ten issues proposed to ensure that all aspects of the ecosystem are accounted for. This would improve the framework through which EcoQOs could be proposed.

The North Sea countries have a long history of promoting an ecosystem approach to fisheries assessment and management. The earliest discussions by ICES on the need for a more formal ecosystem approach to marine fisheries issues were held during the 1975 Symposium on the Long Term Changes in Fish and Fisheries of the North Sea (Hempel, 1975). Countries in the ICES area were also instrumental in the development and implementation of an ecosystem approach to the assessment and management of Antarctic marine resources.

Finally, in 1997 at the Intermediate Ministerial Meeting (IMM) on the Integration of Fisheries and Environmental Issues in the North Sea, the ecosystem approach earned a definitive place on the European policy agenda. The ecosystem approach was especially seen as a concept which could stimulate the integration of fisheries and environmental issues. In the Statement of Conclusions of the IMM, Conclusion 2.6 refers to the ecosystem approach as follows:

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

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

The objective of the North Sea States is to develop a management regime of the North Sea that is based on an ecosystem approach. This approach is considered to be fundamental to achieving sustainable use and protection of the marine environment. The general intention is that management decisions should consider all consequences of human activities for the marine environment in an integrated way. The Oslo Workshop was the follow up of conclusion 2.6 of the IMM in 1997.

The ACME noted that ICES will need to conduct a considerable amount of work on this topic in the near future, as the OSPAR Commission has requested ICES to develop EcoQOs for marine mammals and seabirds in the North Sea in the ICES Work Programme for 2001 from OSPAR.

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19 GLOBAL PROGRAMMES

19.1 Global Ocean Ecosystem Dynamics (GLOBEC) Programme

Request

This is part of continuing ICES work on marine issues.

Source of the information presented

The 2000 report of the Working Group on Cod and Climate Change (WGCCC), a progress report from the ICES GLOBEC coordinator, and ACME deliberations.

Status/background information

The Global Ocean Ecosystem Dynamics (GLOBEC) programme has a number of clearly identifiable elements and products, in particular the Cod and Climate Change programme and its associated Workshops. There are also several GLOBEC-related activities (such as the Working Group on Zooplankton Ecology (WGZE), and the Trans-Atlantic Study of *Calanus finmarchichus* (TASC)).

GLOBEC is the only current international global change programme in which ICES plays a major part, as a regional partner. The plan for future work by WGCCC is included in the International GLOBEC Implementation Plan (IGBP Report 47, 1999). The 2000 WGCCC report includes recommendations for a number of new programme elements in addition to those given in the Implementation Plan. The main focus over the next three to five years will be on synthesis.

The ICES/GLOBEC Newsletters provide further information about progress at regional and national levels and are intended to help with coordination and planning. They are regularly circulated to about 250 scientists and can all be viewed on the ICES website at http://www.ices.dk/newslet/.

The International GLOBEC Newsletter also carries much information about the development of the programme and includes ICES events such as Theme Sessions and Symposia; the address is http://www.pml.ac.uk/globec/.

The future of the ICES/GLOBEC office, and the issue of whether the work which it supports on environment-fisheries interactions should be maintained within the ICES Secretariat, has been under review and discussion for several years.

Information about how environmental variability affects fish stocks is steadily mounting (see, e.g., New Scientist, 20 November 1999, p. 14), as is the evidence that even in the medium term (i.e., 3–10 years) it is unwise to assume that the marine environment will resemble the past mean state. The ACME especially felt that the issue of climate variability and change is of great importance. There are several examples showing how climate variability influences fish population parameters such as recruitment, growth, migration and distribution, and a shift in the climate regime therefore may have dramatic consequences for fish stocks. In order to maintain its prominent role in the scientific investigation of the effects of environmental variability and change on fisheries and the marine ecosystem and the application of new information in this field to marine management, ICES should continue to support GLOBEC-related research. Accordingly, the ACME recommended that ICES continue to host the ICES/GLOBEC Programme Office in order to support the synthesis of work to date within the Cod and Climate Change programme and in fulfillment of the Memorandum of Understanding between SCOR, IGBP, and IOC.

GLOBEC is currently the largest internationally coordinated programme in biological oceanography and will run for at least five more years. ICES has a specific scientific role, as well as a regional role, within the programme and should continue to support relevant research.

Recommendation

The ACME recommends that ICES continue to support GLOBEC-related research and the synthesis of work to date within the Cod and Climate Change Programme.

19.2 Global Ocean Observing System (GOOS)

Request

This is part of continuing ICES work on marine monitoring and assessment.

Source of the information presented

The 2000 report of the Working Group on Shelf Seas Oceanography (WGSSO) and ACME deliberations.

Status/background information

At the 1999 Annual Science Conference, ICES established an ICES/IOC Steering Group on the Global Ocean Observing System (SGGOOS). The activities of SGGOOS include:

1) to develop the ICES-GOOS Implementation Plan including:
    a) development of cooperative arrangements to enhance mutual awareness with IOC and EuroGOOS,

b) development of an ICES Ocean Observing System (I-OOS) based on the ICES Ocean Climate Summary and other relevant products and finding ways to produce and tailor products exploiting the results of the ICES Ocean Observing System,

c) considering the desirability of and possible ways to establish a coordinated and harmonized observation network and designing a system for operational oceanography on an appropriate time scale for the North Sea (in cooperation with EuroGOOS),

d) developing and overseeing the role of the North Sea IBTS quarterly surveys in the Initial Observing System of GOOS, and liaising with and reporting to GOOS bodies as appropriate;

2) to define and promote the role of ICES in GOOS, taking into account input from ICES Advisory and Science Committees;

3) to identify a programme of workshops to facilitate the implementation of ICES-GOOS and to improve awareness of GOOS in ICES, including special sessions at the ICES Annual Science Conference.

ICES should invite representatives of appropriate regional GOOS bodies, such as EuroGOOS, to join the reconstituted Steering Group. Such ICES-EuroGOOS cooperation can be expected to design common plans for the development of operational oceanography to support the management of living resources, coastal areas, and the health of the ocean, and to increase the understanding of climate change.

SGGOOS is scheduled to meet from 23–25 October 2000 at the headquarters of EuroGOOS in Southampton, UK. There is concern that only a few members have been nominated to the Steering Group (ten participants, including the Chairs, from Canada, Germany, Norway, Spain, Sweden, and the UK). Preparations for the meeting have not yet been started, although the development of a flyer designed to stimulate interest in an ICES involvement in GOOS is being contemplated by SGGOOS. It is intended to address the ICES community in general, especially members of Committees and Working Groups not directly affiliated with the Oceanography Committee. It may also serve as a means to encourage organizations such as the North Sea Conference process and OSPAR to support an ICES involvement in GOOS.

At its 2000 meeting, WGSSO reviewed current developments in operational oceanography, especially regional GOOS projects. It was noted that the approach towards an operational system is most highly developed in the Baltic Sea. The Baltic Operational Oceanographic System (BOOS) constitutes a close cooperation between national governmental agencies in the countries surrounding the Baltic Sea responsible for the collection of observations, model operations, and the production of forecasts, services, and information for the marine industry, the public, and other end users. BOOS is being built on existing systems and will develop mainly through commitments from participating agencies. Already most of the components for an operational system are available within national or international programmes. A BOOS Plan 1999–2003 has been published (EuroGOOS, Publication No. 14, January 2000). During this period, BOOS will seek to integrate the existing systems into a uniform entity in order to meet the users' demands for a high quality operational oceanographic service. The BOOS Plan does not make any specific mention of a potential role for ICES.

WGSSO endorsed the view of SGGOOS that there is a need for coordination and harmonization of the national monitoring activities in the North Sea. In particular, it recognized the need to establish a regional ICES-GOOS system focusing on seasonal to decadal time scales for this area. However, it considered that it is important for ICES to work closely with the regional organizations and activities such as EuroGOOS, OSPAR, and the European Environment Agency (EEA) in addition to the governmental institutes in the North Sea countries, which have the responsibility for the long-term environmental monitoring of the area. In order to meet this end, SGGOOS wishes to compile the names and addresses of all the institutes around the North Sea carrying out regular long-term monitoring and operational oceanographic services.

In early 2000, ICES accepted an invitation from EuroGOOS to co-sponsor a Workshop on Bio-ecological Observations in Operational Oceanography. Dr K. Brander, the ICES GLOBEC Coordinator, attended this Workshop, partly because of its relevance to GLOBEC objectives. He gave an opening talk in which he set out the ICES interest in and commitment to GOOS, as articulated particularly in Objective 5a of the ICES Strategic Plan ("Play an active role in the design, implementation, and execution of global and regional research and monitoring programmes").

In spite of the fact that the subjects under discussion were extremely complex, the Workshop was able to identify clear priorities. The Workshop recommendations and summaries of presentations have yet to be published by EuroGOOS. However, amongst the recommendations likely to emerge from the Workshop, the following is of direct concern to ICES:

"An international harmonised, optimised and standardised monitoring of inputs (rivers, atmosphere and adjacent seas) and effects (organisms and ecosystem) of compounds (natural and xenobiotic) to the marine environment should be established with the aim of quantifying and distinguishing natural and man-induced variability. The resultant data should be transformed into indicators for policymakers and end users".

The Workshop made the following observations:

- It was suggested that EuroGOOS and OSPAR put political pressure on national bodies and show examples of the need for the above programme. These examples could be prepared by Working Groups of ICES and designed to show that an international monitoring programme can deliver more information on the state of the marine environment than national programmes.
- An international database must be available making use of modern user interfaces. New data must be put in directly and historical and supporting data must be readily available. ICES could play a major role.
- ICES Working Groups should be responsible for quality criteria for parameters, analytical and statistical methods, processes and model results.

ICES was also invited by GOOS to prepare an article for the eighth edition of the GOOS Newsletter, following the approval by GOOS governing bodies to grant the ICES International Bottom Trawl Survey (IBTS) the status of being a component of the GOOS Initial Observing System.

The International Bottom Trawl Survey (IBTS) is a major survey in the North Sea area. This survey started in the 1960s directed at juvenile herring. Gradually, it was realized that the survey also yielded valuable information for other fish species, such as cod and haddock; the objectives were broadened, and the survey was named the International Young Fish Survey (IYFS). Besides the IYFS, which was carried out in February, a number of national surveys developed in the 1970s and 1980s which were mainly carried out in the summer months. In 1990, ICES decided to combine the international and the national surveys into the International Bottom Trawl Survey. The IBTS was carried out on a quarterly basis from 1991–1996 and twice per year (in February and in August/September) since 1997.

In the early 1980s, a computerized database was set up at the ICES Secretariat. All IBTS data collected from 1983 onwards have been stored in this database. Although the countries participating in this survey were asked to supply the data for the years before 1983, only some have been able to computerize the data for the earlier years of the survey.

The current specific objectives of the IBTS are to:

1) determine the distribution and abundance of the main commercial species with a view to deriving recruitment indices;
2) monitor changes in stocks of commercial fish species independently of commercial fisheries data;
3) monitor stock changes for species which are currently not of commercial interest;
4) collect data for the determination of biological parameters for the more important species;
5) determine the abundance and distribution of late herring larvae (February survey);
6) collect oceanographic data (temperature, salinity, nutrients).

The article explained that because of the systematic, long-term, operational nature of the IBTS surveys and the collection of the resulting data into a common and readily accessible database maintained in the ICES Secretariat, the IBTS is clearly fully consistent with the principles of GOOS. Thus, it qualifies as a component of the GOOS Initial Observing System, albeit as a regional rather than a global entity. These surveys are now seen as a possible core observation system for an ICES GOOS for the North Sea. Identification of other GOOS-relevant observing systems will be undertaken in collaboration with other GOOS-related bodies, such as EuroGOOS.

20 DATA HANDLING

20.1 Activities of the ICES Environmental Data Centre

Request

Item 4 of the 2000 Work Programme from the OSPAR Commission: to carry out data handling activities relating to:

4.1 contaminant concentrations in biota and sediments;

4.2 measurements of biological effects.

Contract from the Helsinki Commission (HELCOM) to serve as Thematic Data Centre for the Cooperative Monitoring in the Baltic Marine Environment (COMBINE) Programme data for a three-year period beginning on 1 July 1998.

Contract from the Arctic Monitoring and Assessment Programme (AMAP) to serve as Thematic Data Centre for the marine component from 1998, extended to 1999 and 2000.

Source of the information presented

Progress report from the ICES Environmental Data Centre and ACME deliberations.

20.1.1 Databases on contaminants in marine media, biological effects of contaminants, and fish diseases

The ICES Environmental Data Bank includes the following components:

1) contaminants in marine invertebrates, fish, birds, and mammals (approximately 330 000 records);
2) contaminants in sea water (approximately 280 000 records);
3) contaminants in marine sediments (approximately 80 000 records);
4) biological effects of contaminants (approximately 4000 records);
5) fish disease prevalence data (approximately 80 000 records).

The annual flow of data into the ICES Environmental Data Centre is approximately 25 000 records for each of the first two components and 5000 records for the third component. The flow of new data into the biological effects database is small. The data holdings according to year are illustrated in Figures 20.1.1 to 20.1.4. It should be noted that the small number of data records from recent years represents a combination of a downward trend in data submissions as well as a backlog in the handling of data submissions at the ICES Environmental Data Centre. No figure has been shown for the annual holdings of data on fish disease prevalence owing to the difficulties of extracting this information in a comparable way from this separate database.

Since mid-1998, ICES has served as the thematic data centre (TDC) for HELCOM under a three-year contract. ICES has been contracted to function as the TDC for the Cooperative Monitoring in the Baltic Marine Environment (COMBINE) Programme (oceanographic, biological, and contaminants) data. This activity, as well as the intended inclusion of older data, is expected to increase the flow of data substantially.

20.1.2 Quality Assurance Database

The ICES Quality Assurance Database consists of the following components:

1) composition of reference materials;
2) written documentation from the laboratories on monitoring activities, analytical methods, etc.;
3) data generated via intercomparison exercises.

The first component currently consists of descriptions of approximately 150 materials with their referenced and/or recommended composition. The second component consists of approximately 250 documents. Countries and/or laboratories reporting data to the ICES Environmental Data Centre are requested to supply additional written information about, e.g., sampling and analytical procedures, but major gaps have been identified in this compilation. The OSPAR procedure has recently been strengthened in this respect and this is expected to result in a more stable flow of information.

The component on intercomparison exercises is based on two major sources of information: the ICES intercomparison exercises and exercises carried out under QUASIMEME I and II. The development of the component based on the ICES intercomparison exercises is hampered by the fact that, in most cases, only paper versions of the reports of the exercises exist. These paper copies have been digitized at the speed that staff resources permit.

Data generated under the QUASIMEME I exercise are confidential, and no solution has been found for the direct access to these data. However, individual laboratories are free to submit these data to ICES as a voluntary contribution. For data generated under QUASIMEME II, a mechanism has been established that allows individual laboratories to report their data to the ICES Environmental Data Centre. Data are being reported by this mechanism.

Figure 20.1.1. Number of records of data on contaminants in biota in the ICES Environmental Data Centre according to year of sampling.

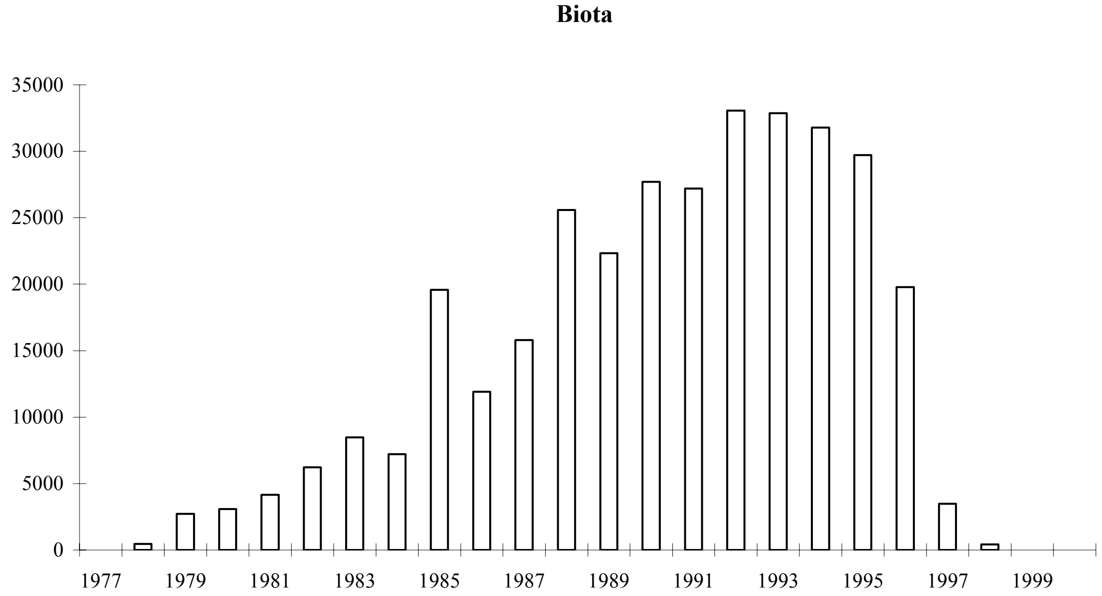


Figure 20.1.2. Number of records of data on contaminants in sea water in the ICES Environmental Data Centre according to year of sampling.

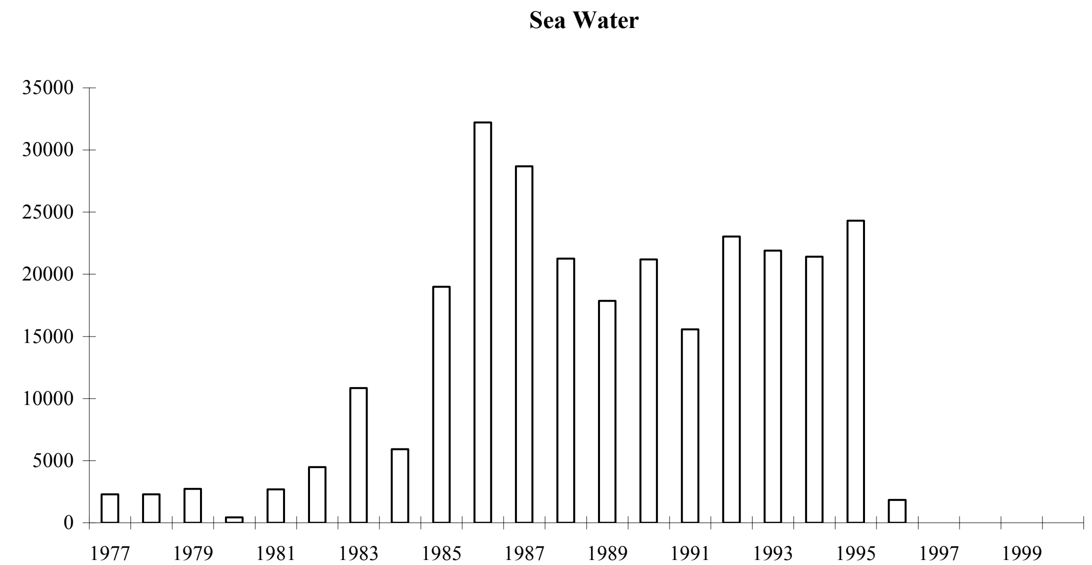


Figure 20.1.3. Number of records of data on contaminants in sediments in the ICES Environmental Data Centre according to year of sampling.

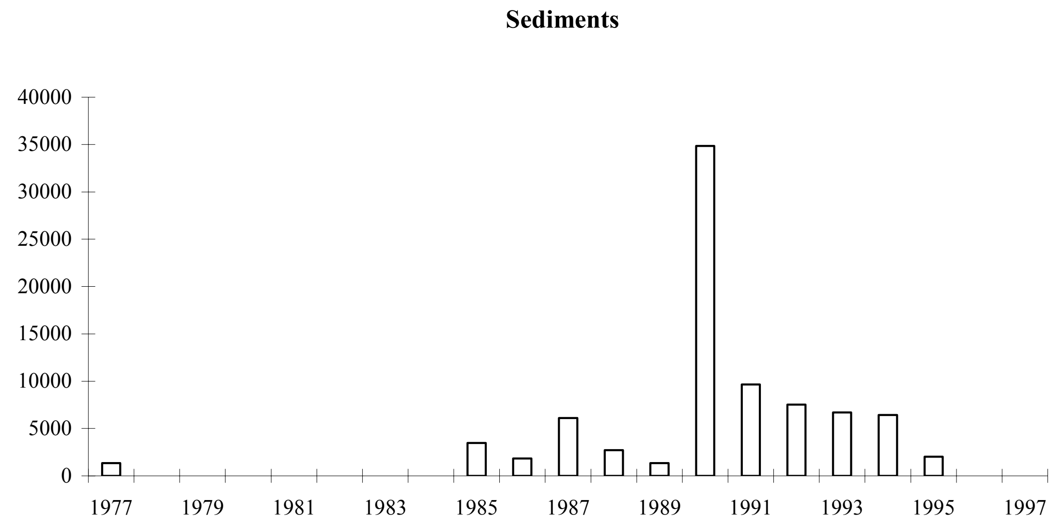
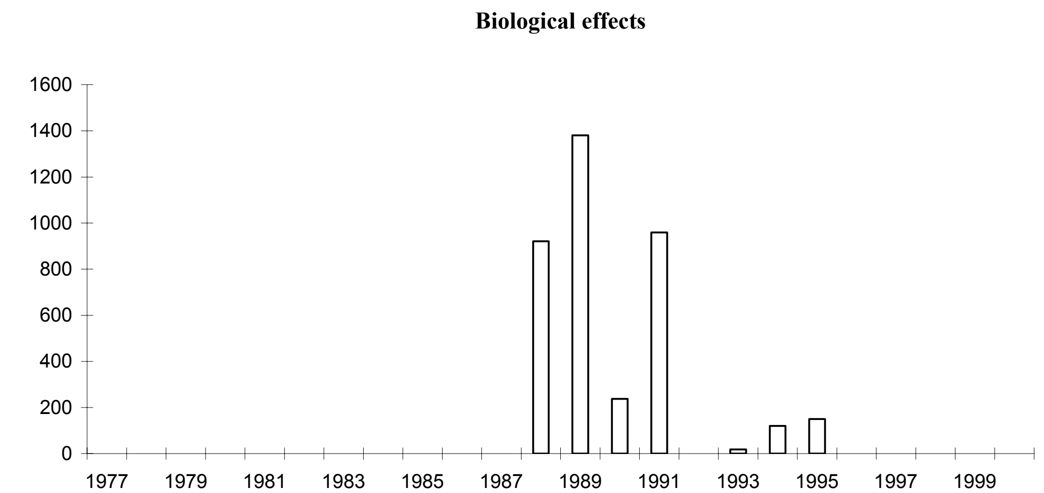


Figure 20.1.4. Number of records of data on biological effects of contaminants in the ICES Environmental Data Centre according to year of sampling.



20.1.3 Handling of data for the Arctic Monitoring and Assessment Programme (AMAP)

ICES has continued to serve as the Thematic Data Centre for the marine component of AMAP under a 1998 contact that has been prolonged to cover 1999 and 2000 as well.

20.1.4 European Environment Agency (EEA)

The ICES Environmental Data Centre has processed requests for the provision of data on contaminants in biota and sediments for a project (EUMARIS GIS) being carried out by the European Topic Centre for the Marine and Coastal Environment.

20.1.5 Major data products

The ICES Environmental Data Centre provided information for the report "Status of fisheries and related environment in Northern Seas" prepared by ICES for the Nordic Council of Ministers (see Section 4, above). Apart from data, the information supplied also included plots and univariate statistics on concentrations of cadmium and ΣDDT in cod liver from the Baltic Sea, Kattegat, North Sea, and the Norwegian Sea.

A number of data sets on various contaminants in herring, flounder, and blue mussel from the Baltic Sea were supplied for use in the preparation of a chapter on contaminants in Baltic marine biota for the HELCOM Fourth Periodic Assessment of the State of the Baltic Marine Environment, 1994–1998.

20.1.6 On-line access to data inventories

Access to dynamic/query inventories of the data held in the ICES Environmental Data Centre can be established through an interface on the ICES website [http://www.ices.dk/env]. The interface allows the user to retrieve information about present holdings of contaminants and biological effects data. It should be noted, however, that not all the data in the ICES Environmental Data Centre are visualized by this web interface as some laboratories have not given permission for their data to be included in this inventory. In the future, it is intended that information on all environmental data submitted to ICES will be included in this inventory.

20.1.7 Fish Disease Database

Several submissions have been received for this database in 1999/2000.

20.1.8 Marine Mammals By-catch Database

No new data were submitted to the database in 1999/2000.

20.1.9 CoastBase

ICES is a partner in the CoastBase project which is funded by the EC within the Fifth Framework Programme. The aim of the CoastBase project is to increase the accessibility of data and metadata in order to improve policy-making and cooperation along the European coasts by developing a prototype of a data warehouse containing relevant information. Data products from both the Environmental and the Oceanographic Data Centres are being considered for inclusion in this prototype.

20.1.10 ICES Environmental Data Reporting Format

As reported last year, there still seems to be a difference of opinion with regard to the usefulness of the ICES Environmental Data Reporting Format when used as an exchange format. Some laboratories have developed their own software for the transfer of data from their own database to ASCII files, as specified in the ICES Environmental Data Reporting Format, and these laboratories favour that the data exchange format should be maintained as it is, while other laboratories suggested that it should be possible to submit data by other means, e.g., as an EXCEL spreadsheet. The ICES Environmental Data Centre will review potential alternatives for data submission that would meet the needs of institutes that experience difficulties with the present data exchange format. It is important, however, to emphasize that this development will demand resources for programming and implementation.

Hopefully, the development of a user-friendly means of data submission will encourage the laboratories to submit more data to the ICES Environmental Data Centre.

Recommendations

The ACME recommends that the development of another exchange format (spreadsheet format) should be given high priority, as this is expected to encourage more laboratories to submit the large amounts of data that are stored in national databases to the ICES Environmental Data Centre.

20.2 Handling of Nutrient Data for the OSPAR Commission

Request

Item 3 of the 2000 Work Programme from the OSPAR Commission: provision of advice and standard data products to a small group of OSPAR experts (3–5) working at the ICES Secretariat for the purposes of developing the Common Procedure for the Identification of the Eutrophication Status of the Maritime Area and contributing to further NEUT intersessional work.

Item 4.3 of the 2000 Work Programme from the OSPAR Commission: to carry out data handling activities relating to the implementation of the Nutrient Monitoring Programme.

Source of the information presented

Progress report from the ICES Oceanographic Data Centre and ACME deliberations.

Status/background information

Activity has been very low in the past twelve months, partly due to the lack of resources to provide updated inventories, and partly because very few nutrient data were received that were specifically identified as having been submitted for OSPAR uses. Indeed the only activity has been the handling of the Belgian (MUMM) 1998 data set. Some research nutrient data sets have been received, but not all have been dealt with up to one year after submission to the Secretariat.

The OSPAR Working Group on Nutrients and Eutrophication (NEUT) met in London in October 1999 at which time it reviewed the nutrient data handling sections in the 1998 and 1999 ACME reports. NEUT concluded the following from its deliberations:

a) The German delegation at NEUT felt that it was not clear from the information provided why nutrient data reporting levels were high in the early 1990s, and that this may be an artifact of mis-assignment of cruise data. ICES should therefore provide more information in the future on data submissions. The German delegation would nevertheless check with national institutes to determine if the decline in data submissions was real.

b) It was possible that part of the decline in submissions of nutrients data to ICES may be owing to difficulties experienced by OSPAR Contracting Parties in using the ICES databases.

c) There was a general request for ICES to present in future more detailed information to NEUT regarding the OSPAR nutrient data held at ICES.

d) OSPAR Contracting Parties should contact institutes submitting nutrients data to ICES to ensure that they flagged OSPAR nutrient data in their submission to ICES.

e) On the subject of arrangements for taking forward the request for advice and assistance regarding eutrophication issues, Germany and the Netherlands were requested to make direct contacts with the ICES Oceanographer to clarify how, where, and when this work should be carried out to fulfill the work requested in item 3 of the 2000 Work Programme from OSPAR.

The ACME was informed that there has been very little follow up with regard to any of the above items. The following observations were made by the ICES Oceanographer with regard to the above-mentioned issues:

a) This comment seems to arise from a misunderstanding as the information presented was based, as it always is, on the whole ICES nutrient database. Mis-assignment is not the issue. Very few nutrient data supplied to ICES are identified as OSPAR data on ROSCOP forms. Whenever information is so received, data are appropriately flagged, as with all ICES project data sets. With regard to the decline, Germany has written to confirm that the decrease is real from their perspective, but other evidence does not confirm this to be generally the case.

b) OSPAR has been unable to clarify the basis of this statement which, if true, should be considered as a serious problem.

c) The information presented to the ACME is, for practical reasons, brief. Normally very detailed information is provided directly to NEUT, but on this occasion it was not possible to submit information because of the very short notice provided. This situation was exacerbated by the fact that the Secretariat was unable to be involved in this meeting due to lack of resources, and its proximity to the 1999 ICES Annual Science Conference.

d) There has been no outcome of this request. It has frequently been requested that ROSCOP forms should additionally contain this information, but only two countries (Germany and Belgium) regularly adhere to this.

e) There has been no follow up of the request that ICES assist a small group of OSPAR experts working on NEUT intersessional work. It is assumed that this group will build on the proposed analysis of nutrient data described in the 1997 ACME report (ICES, 1997). However, NEUT did not react on this at its October 1999 meeting.

The ACME took note of the above information and agreed that the issue of nutrients and eutrophication should be approached on a broad basis.

Reference

ICES. 1997. Report of the ICES Advisory Committee on the Marine Environment, 1997. ICES Cooperative Research Report, 222: 115, 185–187.

20.3 Development of Biological Databases

Request

Item 5 of the 2000 Work Programme from the OSPAR Commission: to continue to establish data banks for phytobenthos, zoobenthos, and phytoplankton species.

Contract from HELCOM to serve as the Thematic Data Centre for COMBINE Programme data for a three-year period beginning on 1 July 1998.

Source of the information presented

Progress report from the ICES Environmental Data Centre and ACME deliberations.

Status/background information

In 1997, OSPAR and HELCOM requested ICES to make preparations for establishing databases on phytoplankton, zooplankton, phytobenthos, and zoobenthos. Specifically, the Commissions have requested ICES to prepare reporting formats and data entry/data-screening software for data submissions on these parameters. The work is coordinated from the ICES Secretariat, but has also been based on input from several Working Groups and individual scientists.

In February 2000, the ICES/OSPAR Steering Group on Quality Assurance of Biological Measurements related to Eutrophication Effects (SGQAE) and the ICES/HELCOM Steering Group on Quality Assurance of Biological Measurements in the Baltic Sea (SGQAB) reviewed the final draft of the ICES Biological Data Reporting Format. These groups made several suggestions for additions and amendments to the reporting format and these suggestions have been incorporated into the reporting format as far as possible. The Biological Data Reporting Format is available on the ICES website at http://www.ices.dk/env/.

A data entry program that facilitates the entry of data into the ICES Biological Database is under development. This data entry program will be linked to specific code lists as, for example, a species list/species code list in order to secure the use of a correct nomenclature of the species. Apart from the entry of raw data, it is the intention, as for the contaminants data, that a facility should be developed in order to be able to transform data from national and regional database structures to the ICES Biological Data Reporting Format structure.

The ACME was informed that development of the Biological Community Database has been seriously delayed owing to several circumstances.

Noting the above, the ACME emphasized the importance of finalizing the Biological Community Database as soon as possible. The ACME further recommended that the development of a database on non-indigenous species should take place within the framework of this database.

20.4 ICES Phytoplankton Checklist

Request

This is part of ongoing ICES work related to phytoplankton ecology issues and coordination of quality assurance issues and data handling with regard to measurements of phytoplankton.

Source of the information presented

Report of the Study Group on an ICES/IOC Checklist of Phytoplankton (SGPHYT) and ACME deliberations.

Status/background information

The ACME was informed that, during the meeting of the Working Group on Phytoplankton Ecology (WGPE) in April 2000, the criteria and format of a checklist were discussed. It was concluded that the identification of phytoplankton species is in a state where optical techniques and molecular probes are rapidly developing. At the same time, it was stressed that there is continuously an obvious need for knowledge of species identification from microscopic analysis.

Taxa usually included in the phytoplankton checklists and analysed as phytoplankton also cover other groups of organisms. Heterotrophic microplankton and cyanobacteria are examples of this. The amount of information that is desired in a list of this kind is so large that a database is needed for data storage and updating. The ACME suggested use of the working name “Relational Database on Microplankton Protists”. Although a considerable amount of information, including pictures and sizes of the species, is suggested to be included in the database, it is not intended to be used for species identification.

At present, a questionnaire is being compiled, in which phytoplanktologists are asked to give their opinion on specific items of information to be included in the list. It is anticipated that the criteria and format of the list will be ready to be presented at the 2000 ICES Annual Science Conference. At that time, the “full” list of existing checklists will also be presented. Thereafter, the work on the compilation of the new complete list will start. The first step will be checking that the species meet the criteria to be included and are taxonomically sound. This work will initially be done by correspondence, but at a later stage workshops are anticipated.

The ACME endorsed this work on the development of a checklist and, ultimately, a database.

20.5 Development of Reporting Format for Biological Effects Measurement Data

Request

Item 6 of the 2000 Work Programme from the OSPAR Commission, stated in detail in the text below.

Source of the information presented

The 2000 reports of the Working Group on Biological Effects of Contaminants (WGBEC) and the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO), a progress report from the ICES Environmental Data Centre, and ACME deliberations.

Status/background information

As part of the work programme for OSPAR, ICES was requested to expand the Environmental Data Reporting Format to include all the reporting parameters required for each of the biological effects techniques adopted by OSPAR, where these reporting formats have not already been developed. In so doing, this work was to be undertaken in three stages:

Stage 1: the development of reporting formats for:

- P4501A1 (EROD);
- PAH metabolites;
- DNA adducts;
- Liver histopathology;
- Liver nodules;
- TBT (intersex, imposex).

Stage 2: the development of reporting formats for:

- Metallothionein;
- ALA-D;
- Bioassays;
- Fish reproductive success.

Stage 3: the development of reporting formats for:

- TBT shell thickening;
- Lysosomal stability;
- Antioxidant enzymes.

Most of the work on this request will be conducted later in 2000, as information on quality assurance requirements for the above-mentioned methods is needed from the Biological Effects Quality Assurance in Monitoring Programmes (BEQUALM) project (see Section 7.3, above). It is anticipated that information from this project will become available before the end of 2000.

The different measurements of biological effects of contaminants described and discussed in the monitoring guidelines of OSPAR for the Coordinated Environmental Monitoring Programme (CEMP) and at the 1999 WGBEC meeting have been compiled. Furthermore, the relevant volumes of the *ICES Techniques in Marine Environmental Sciences* series have been reviewed in order to check details of the relevant measurements. These measurements have been compared with the existing ICES Environmental Data Reporting Formats and the relevant ICES codes, and a series of tables was produced.

WGBEC was asked to consider whether the listing of measurements in this series of tables is appropriate for inclusion in the reporting format. Furthermore, WGBEC was requested to consider whether any relevant and applicable measurement, data parameter, or metadata is lacking on this listing. At its 2000 meeting, WGBEC discussed this information on reporting formats supplied by ICES. There was general agreement that some properties will be similar for all biomarkers. These properties can be found in Table A10.1 in Annex 10. In addition, WGBEC agreed that protein measurements should be kept separate from other biomarkers although they would obviously need to be linked to the appropriate biomarker measurement (see Table A10.2 in Annex 10). The specific methods that need attention are listed in Tables A10.3 to A10.7 of Annex 10.

WGBEC noted that the formats for imposex at present can only accommodate results for populations. For future needs and flexibility, WGBEC suggested that the structure be changed so that results for individual gastropods are entered. All aggregate results can then be calculated from this database. Furthermore, it is necessary that the entries be modified so that additional gastropods to *Nucella* can be used. At present, *Buccinum undatum*, *Littorina littorea*, *Hinia (Nassarius) reticulatus*, and *Neptunia* spp. should also be included. If the current structure is retained, the frequency of affected female gastropods is an additional field that should be included. For this reason, no table for imposex/intersex measurements has been included in Annex 10.

WGBEC emphasized that it is vital that information on individual fish can be linked so that all information concerning one individual or pooled sample, including histopathology, contaminant concentrations, fish morphometry, and biomarker responses, can be retrieved. Bioassay results must be linkable to data on the relevant matrix.

It was pointed out that some methods may need to be modified following the outcome of the BEQUALM QA/QC programme.

The ACME noted that the new types of measurements approved by WGBEC will be included in the version of the ICES Environmental Data Reporting Format that is expected to be issued in late 2000 or early 2001.

As reporting requirements for liver histopathology data are also part of this request, at its 2000 meeting, the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) considered changes needed in the Environmental Data Reporting Format with regard to the incorporation of liver histopathology data. Based on the results of the 1999 BEQUALM Weymouth Workshop on External Fish Diseases and Liver Histopathology (see Section 7.3, above), WGPDMO recommended the addition of a number of new disease categories to the existing ICES Fish Disease Data Reporting Format, which were selected on the basis of their importance to indicate contaminant exposure. These disease categories are:

- hepatocellular and nuclear polymorphism;
- hydropic vacuolization of biliary epithelial cells and/or hepatocytes;
- phospholipidosis;
- fibrillar inclusions;
- spongiosis hepatis (known from North American flatfish species);
- clear cell foci;
- vacuolated foci;
- eosinophilic foci;
- basophilic foci;
- (mixed foci will be allocated to one of the above categories according to the predominating cell type);
- hepatocellular adenoma;
- cholangioma;
- pancreatic acinar cell adenoma;
- hemangioma;
- other benign tumours;
- hepatocellular carcinoma;
- cholangiocarcinoma;
- mixed hepatobiliary carcinoma;
- pancreatic acinar carcinoma;
- hemangiosarcoma;
- hemangiopericytic sarcoma;
- other malignant tumours.

As to the non-specific liver lesions, a consensus on the types of lesions to be included in the data bank could not be reached. It was therefore agreed to postpone a decision and to utilize the experience regarding diagnostic criteria and accuracy to be gained in the forthcoming BEQUALM ring tests.

Originally, WGPDMO had recommended the incorporation of only five disease categories in the reporting format (ICES, 1998). However, after having reviewed the results of the BEQUALM workshop, WGPDMO considered it advantageous to use more than five categories based on specific lesions since that would add considerably more information to the ICES disease database, enabling a more detailed statistical analysis, e.g., with a view to establishing time scales and predictions for the development of neoplastic lesions on a regional basis as part of risk assessment. A grouping of data according to the original five categories could still be done if considered appropriate.

WGPDMO recommended that the ICES Fish Disease Data Reporting Format should be modified according to the above categories of liver lesions and that the lesions should be added to the existing list of externally visible diseases. It was emphasized, however, that the revision of the ICES Reporting Formats should ideally be postponed until the BEQUALM project is finalized, since data will only be submitted to ICES after the quality assurance procedures for the diagnosis of histopathological liver lesions have been established successfully. It cannot be excluded that the experience gained in the course of the project might possibly lead to a need for a further revision of the list of histopathological liver lesions to be reported to ICES.

Reference

ICES. 1998. Report of the Working Group on Pathology and Diseases of Marine Organisms. ICES CM 1998/F:4.

ANNEX 1

SYNTHESIS OF THE CURRENT KNOWLEDGE ON NORMALIZATION TECHNIQUES FOR CONTAMINANTS IN SEDIMENTS

1 TEMPORAL TREND SURVEYS

1.1 Introduction

Many countries already have temporal trend programmes in place for contaminants in sediments. These may have been established on the basis of existing Guidelines, or else designed prior to the existence of guidance from OSPAR or ICES. In other cases, countries may wish to establish new temporal trend programmes on the basis of best current advice. WGMS therefore considered how these two situations should best be accommodated. It was agreed that any new Guidelines should not automatically lead to alterations in the protocols for existing temporal trend studies. A significant proportion of the value of temporal trend programmes is their length and the consistency (sampling techniques, analytical methods, data manipulation, etc.) with which they have been undertaken. It was therefore recommended that existing programmes should continue. However, the statistical power of such programmes to detect changes should also be established, and the usefulness of the programmes assessed against the needs of the project. If the power is considered adequate, then no changes need be made. If the power is not adequate, then it may be advisable to adopt improved survey designs to increase the power. If it is proposed to change the design of the programme, then it may well be appropriate to undertake the survey using the two different designs in parallel for a sufficient number of years to establish the relationship between the data from the two strategies, and thereby preserve the temporal length of the data set.

1.2 Grain Size Fraction

WGMS then considered the design of new temporal trend surveys for selected trace metals, PCBs, and PAHs. Current Guidelines stipulate that a fine fraction should be separated from the sediment prior to analysis. WGMS recommended that initial consideration should be given to the particular circumstances of the monitoring location and the behaviour of the target contaminant. In most locations, the contaminants of interest would be concentrated in the fine-grained material. Also, it is likely that the fine-grained material is more actively involved in exchange processes between solid and aqueous phases, and therefore may give some indication of the availability of the contaminants.

It is therefore recommended that, in general, analyses should be carried out on a sieved fine fraction. In most circumstances, the < 63 µm fraction will be suitable, although separation of a finer fraction, for example, < 20 µm should further reduce the residual variance.

WGMS was aware that some circumstances exist where contaminants may be present in high concentrations in coarser materials (e.g., particulate waste from some mining activities). In such cases, the participating laboratory may prefer to analyse a coarser fraction. This would be acceptable, as the main purpose of temporal trend studies is to make comparisons over time within a series of independent locations. The critical factor is to ensure that once a size fraction is selected for a location, this fraction is clearly recorded in the programme design for that location, and that the fraction does not vary from year to year.

1.3 Need for Normalization

In the context of temporal trend surveys, normalization is a process that reduces the variance of the data set by taking account of differences in grain size distribution and mineralogy (gross sediment composition) between samples. Both the selected trace metals and the organic contaminants will co-vary strongly with such factors and grain size, clay mineral and organic carbon content. Differences in these co-factors between samples will often be reflected in differences in the concentrations of the target contaminants.

The separation of a defined grain size fraction will greatly reduce that component of the variance between samples that arises from differences in grain size distributions. However, it is known that differences will usually remain between samples in mineralogy and organic carbon content. It is therefore recommended that appropriate co-factors be measured in the selected grain size fraction in association with the contaminant concentrations.

For CBs and PAHs, the most appropriate co-factor is the organic carbon concentration. This should be measured either by a CHN elemental analyser or by a wet oxidation method. It is recognized that both of these methods can provide accurate and reliable data. However, a review carried out by WGMS in 1994/1995 (summarized in ICES, 1994) indicated that elemental analysis was preferable as it was subject to fewer potential interferences. Reference materials for organic carbon concentrations are readily obtainable, and the determination is included in the scope of QUASIMEME.

For Cd and Pb, the most appropriate co-factors are organic carbon and clay content. The clay content is not easily measured with the necessary reliability. Therefore, surrogate measures of the concentrations of elements that are largely confined to clay minerals in fine sediment should be employed, for example, aluminium or lithium concentrations. Other elements such as scandium may also be suitable. The analytical method chosen for these determinations should be designed to measure the total concentrations of the metallic co-factors. In fine fractions, concentrations of these elements should be measured either by total digestion (for example, using HF) or by a thorough partial digestion involving nitric acid. Aqua regia has been shown to not always be suitable, as recoveries of aluminium are often low. The same digestion process should be used for the metallic co-factors as is used for the Cd and Pb determinations. Reference materials for aluminium and lithium concentrations (by total digestion) are readily obtainable, and both determinations are included in the scope of QUASIMEME. Reference materials for partial digestions are much less available.

For Hg, the same comments as made above in relation to Cd and Pb also apply, with the additional comment that care should be taken to ensure that the digestion mixture remains sufficiently oxidizing to prevent the loss of mercury. For example, a total digestion mixture should include both HF and nitric acid.

1.4 Effectiveness of Normalization

As indicated above, the purpose of normalization is to reduce the overall variance of a data set about any fitted temporal trend, and thereby increase the statistical power of the programme. In order to select the most appropriate normalizer for any series of temporal trend data for Hg, Cd or Pb, the concentrations of the contaminants of interest should be expressed as ratios to the concentrations of the co-factors. The magnitude of the residual variances should then be compared and the most effective normalizer (i.e., that which results in the lowest residual variance) selected.

Data for the concentrations of a range of co-factors covering clay mineral content and organic matter (as a minimum, organic carbon, plus aluminium or lithium) should be collected on each sampling occasion. The selection of the most effective normalizer for each series of data should be checked at each data assessment. It is not necessary to require that the same normalizer be used for every temporal trend series. It is not necessary that the same normalizer be used on each occasion that the data from any individual location are assessed.

The above discussion has assumed that the distribution of the contaminants of interest is largely controlled by the distribution of organic matter and clay minerals. Circumstances exist where this is not the case, for example, in some sediments rich in calcium carbonate the majority of the cadmium may be held in shell fragments. In such specialized circumstances, it may be more appropriate to utilize a co-factor that reflects that component of the sediment which has the greatest influence on the distribution of the contaminant, for example, calcium concentrations to reflect shell fragments.

2 SPATIAL SURVEYS

2.1 Introduction

Coordinated international surveys of the spatial distribution of contaminants in sediments have been carried out on irregular occasions, and the current OSPAR strategy for the Coordinated Environmental Monitoring Programme (CEMP) contains a similar provision. Temporal trend studies will be a continuing activity, while spatial distribution surveys will only be carried out when a particular need arises.

The last major assessment of spatial distributions of contaminants in sediments was carried out by the OSPAR Ad Hoc Working Group on Monitoring (MON) in 1992 in association with the preparation for the 1993 North Sea Quality Status Report. The data available for assessment comprised analyses of whole sediment for organic contaminants, and of the total concentrations of inorganic contaminants in whole sediment. The sampling points were irregularly distributed across the study region, and the spatial density of samples varied greatly from area to area.

It proved difficult to derive reliable descriptions of the distribution of organic contaminants, normalized to organic carbon, because of the limited number of sampling points and uncertainties over the quality of some of the data. The assessment of the metals data was more successful, although similar difficulties were encountered. The metals data were normalized to aluminium by taking a simple ratio between the two determinands. There was unacceptable uncertainty in the normalized values in many sandy sediments, and the assessment group found it necessary to eliminate all sediment samples that contained less than 1 % aluminium.

This experience formed the background to the WGMS discussion, which also drew on the report of the ICES Study Group on Monitoring Programmes for Contaminants in Sediments (ICES, 1997).

2.2 Sampling Locations

The fundamental output from an exercise designed to map the spatial distribution of contaminants in sediments is a representation of the geographical variations of concentrations of the contaminants over the area of interest. This may be contrasted with exercises designed to describe the "average" concentration of contaminants in an area, which seek to summarize the concentrations (for example, as a mean, possibly with an expression of variance) over an area, and are not concerned with the geographical patterns of levels within the area. The basic data set for a mapping exercise will consist of measurements of geographical patterns of levels in sediments at a number of locations over the sampling area. The geographical distribution of the values, e.g., displayed by plotting the values on a base map, is the simplest form of display of the data.

However, the purpose of mapping is normally to estimate the levels of contaminants at locations where samples have not been collected, for example, at locations between sampling points. There are wide ranges of interpolation procedures that can be used to estimate the levels at locations that have not been sampled. A common subsequent step is to draw contour lines joining locations of equal levels (measured or estimated) of contaminants.

The details of the most appropriate sample distribution schemes depend to some extent upon the statistical techniques and models to be used to generate estimated values, and to draw the subsequent contours. However, in general, a regular sampling grid (triangular or square), supplemented by a lesser number of more closely spaced sampling locations, provides a good basis for the preparation of maps.

2.3 Analytical Strategy

WGMS recognized that several different analytical strategies have been discussed for spatial surveys of contaminants in sediments. These include the collection of data representing the total concentrations of contaminants in whole sediment, and the analysis of a defined size fraction. In each case, there are ranges of subsequent forms of data manipulation that can be employed prior to the presentation of the final data products (maps). The natures of the basic data, and of the subsequent data manipulations/normalization, determine the nature of the final output, and therefore the objectives that can be met. It was decided that it would be appropriate to review the main applications (and limitations) of a range of analytical and manipulation strategies.

2.4 Trace Metals—Mercury, Cadmium, and Lead

2.4.1 Total digestion of whole sediment

Without normalization

The determination of the total concentrations of contaminants in whole sediment provides a direct measurement of the concentrations that are found in the sediment at the sampling locations (although material coarser than 2 mm is normally excluded). Maps can be drawn utilizing these data, which give a direct reflection of the conditions (contaminant concentrations) in the seabed sediments.

Experience has shown that the geographical patterns that emerge are closely related to the patterns of sediment grain size distribution (distribution of fine-grained material) over the study area. This arises because, in general, the contaminants of interest are concentrated in the fine-grained material. Differences that may arise from other factors, such as anthropogenic inputs, may therefore be obscured by variations in grain size.

With normalization

It is possible to apply geochemical normalization procedures to these data. The most commonly used procedure is to express the metal concentrations as ratios to the concentration of aluminium or lithium, i.e., to normalize for variations in grain size as reflected by aluminium and lithium, components of the clay minerals in the sediments. Experience has shown that this may reduce the variance of much of the data, but will probably increase the variance in sandy sediments. This occurs for several reasons: firstly, because the precision of the data for both contaminants and normalizers is worse at the low concentrations that are found in sands and, secondly, because the co-factors aluminium and lithium are not necessarily confined to the clays. Minerals such as feldspars, which contain aluminium, are found in the sand-grade material. The effect is to give non-zero intercepts to regressions between contaminants and normalizers. The distortion of the ratios caused by these intercepts becomes progressively worse at low concentrations of normalizers. This distortion cannot be fully rectified by improvements in the quality of the analyses.

Maps of normalized concentrations calculated in this way have been found to be misleading if all samples are included. The maps are much less misleading if samples containing aluminium concentrations below a certain threshold, e.g., 1–2 %, are eliminated from the data set. It has been shown to be possible to identify areas of anomalous concentrations, particularly if the normalizer concentration is high. It is not possible to identify such areas in sandy sediments, unless the anomaly is large (e.g., enhancement of concentrations by a factor of 2 or more).

It might be possible to improve the sensitivity of this procedure if the value of the intercept was known accurately, and its variation over the survey area was also known. Mathematical procedures could then be used to subtract the value of the intercept and calculate contaminant to co-factor ratios based upon data corrected for the intercept. However, good values for the intercepts are not currently available, and derivation of such values would require considerable effort.

Correction of analyses to a sediment of standard composition

Rather than express normalized concentrations as a ratio to the normalizer, an alternative approach has, at times, been adopted. This seeks to express the contaminant concentration as the equivalent concentration that would be predicted to be present in sediment of a standard composition (e.g., containing 25 % material less than 63 μm). The most common procedure is to undertake a whole sediment analysis for the contaminants, and also to undertake a partial grain size analysis to determine the proportion of the sediment finer than a chosen grain size, e.g., 63 µm.

The data can then be processed in one of two ways:

Corrected conc. = [(Measured concentration) / (percent fines)] × (standard % fines)

This procedure assumes that the intercept of the regression of the contaminant concentration against the percent fines is zero. It will give anomalous results if the intercept is not zero.

Corrected conc. = ([(Measured conc. – conc. in coarse fraction) / (percent fines)] × (standard % fines)) + (conc. in coarse fraction)

The corrected concentrations obtained by this method could be used to draw maps. The calculations require accurate data on the concentrations of contaminants in the coarse fraction, and also for the amount of fines in the sample. Both of these measurements can be subject to considerable uncertainty, and both are of particular significance in sandy sediments. This form of data manipulation therefore suffers from the same sources of errors as discussed in the preceding section on geochemical normalization. It is likely that this method could be effective in sediments containing a high proportion of fine material, but would be much less reliable in sandy sediments. In order to make reliable interpretations of maps, it may be necessary to exclude samples with less than, e.g., 5 % or 10 % fine material, depending on the acceptable propagated error in the normalized value.

Methods based on regressions

A number of methods of data analysis have been used which rely upon the ability to draw regression lines between contaminant and normalizer concentrations for the area of study. A fundamental requirement is therefore that a sufficient number of samples is collected within an "area" covering a wide range of sediment types (sand to mud). This could lead to a requirement to analyse many samples to cover large marine regions, considered in a number of different "areas".

The simplest procedures compare the gradients of regressions drawn for samples from different areas. The steeper the gradient, the more contaminated an area is considered to be. This has been shown to provide useful perspectives on superficially similar areas, such as a series of fjords, but does not provide information suitable for mapping.

Other procedures involve the calculation of residual contaminant concentrations about a fitted regression line. Positive residuals indicate that the concentrations are greater than would be predicted from the concentration of normalizer in that sample. Negative residuals indicate a lesser contaminant concentration than predicted. Statistical procedures can be used to indicate which samples are statistically different from the predicted values, and therefore may indicate some additional source of contaminant at that particular location. Maps drawn on this basis have successfully identified contaminated areas of northeast England, in the Irish Sea, and elsewhere.

A significant problem with this approach is the uncertainty as to whether the regression lines represent natural uncontaminated conditions, and positive residuals therefore represent contamination, or whether the regression line itself may reflect contaminated conditions, and positive residuals reflect additional contamination. The presence of points with significant negative residuals may support the latter suggestion. The method cannot therefore be used reliably in a quantitative manner, i.e., to compare the true degree of contamination at different locations.

A development of this procedure relies upon the identification of background concentrations of contaminants and co-factors. This can be approached through sediment cores in suitable locations, or through geological sampling of Quaternary sediment. If it is possible to establish true background values for the regression lines, then positive residuals may be used to quantitatively reflect contamination. Recent research suggests that metal to aluminium ratios have been relatively constant through the Quaternary Period over a large part of northern Europe. Considerable further work is required to establish background values over larger areas. However, the procedure has the potential to quantify the degree to which sediments are contaminated in comparison to "historic" values. It is likely that maps drawn on this basis would still contain some grain size effect, and procedures to take this into account would need to be developed.

2.4.2 Partial digestion of whole sediment

Partial digestion refers to digestion methods which do not result in total dissolution of the sediment sample, and which therefore may not measure the total concentrations of contaminants or co-factors in the sediment. Methods of this nature have proved difficult to standardize. Differences in mineralogy between samples may require alterations to be made to digestion procedures to obtain "comparable" degrees of decomposition. Certified reference materials (CRMs) for partial digestions are not commonly available.

Partial digestion of whole sediment is not recommended for coordinated international programmes on a broad geographical scale.

2.4.3 Analysis of sieved fine fraction of sediment

Analysis of fine-grained material sieved out of sediment can be used to provide data for maps of the distribution of contaminants in fine-grained material over wide areas. The separation of a fine-grained fraction prior to analysis is a direct and effective way to "normalize" for grain size differences between samples. The material analysed will usually contain both contaminants and co-factors in relatively high concentrations. The analytical task is therefore simplified as most measurements can be made well above analytical detection limits, in concentration ranges where accuracy and precision are relatively good. The fine-grained material is likely to be more actively involved than coarse material in exchange processes with sea water, and also to be more closely associated with marine macrofauna.

Recent studies in North Sea estuaries have shown that there can be considerable differences in the bulk composition of fine-grained fractions between areas. In some areas, the fraction may be dominated by clay minerals, and in others there may be a significant component of fine silica. However, these studies have also shown that the concentration of clay minerals is well correlated with the concentration of aluminium or lithium, with low intercepts. Simple normalization procedures (ratio of contaminant to normalizer concentrations) can therefore be applied effectively. Ratios of aluminium or lithium to clay mineral content vary by 30–50 % over the North Sea, reflecting differences in clay mineralogy, and suggesting that simple normalization will be adequately effective over wide areas.

The separation of the fine fraction from the bulk sediment is an additional step required of the analyst, and care is required to ensure that contamination does not occur. However, complete separation of a fine fraction from the bulk sediment is not necessary, as current information indicates that, once normalized, concentrations do not vary significantly between 20 μm and 63 μm.

It is therefore recommended that the analysis of a fine fraction and mapping of normalized concentrations should provide the most reliable method to compare concentrations over wide areas, e.g., the North Sea. There is currently insufficient information to assess the effectiveness of the method over the whole OSPAR or ICES areas, but theoretical considerations suggest that it will be more reliable than the other procedures discussed above. This procedure does not describe the whole sediment, but describes a part of it, namely, the fine-grained component. Separation, analysis, and normalization of this component should provide a consistent basis for a reliable and comparable description of contaminants in sediments.

There are theoretical reasons to favour the analysis of a fine-grained fraction (such as material < 20 μm) rather than a coarser fine fraction (such as < 63 μm). However, the separation of this fraction is considerably more laborious than separation of the < 63 μm fraction, and the degradation in quality of data that would result from selection of the coarser fine fraction would probably be outweighed by the increased analytical output. In either case, the processing of large numbers of samples, particularly samples containing low amounts of fine material, would be greatly facilitated by the use of an automatic recirculating sieving machine.

It has been shown that the results of analysis of the fine fraction by either total digestion (HF) or strong partial digestions (nitric acid) are not significantly different. Both acids decompose the clays, where both the elements of interest and the co-factors (aluminium and lithium) are concentrated. The use of aqua regia is not recommended, as recoveries of aluminium are poor. The adoption of a method designed to determine the total concentrations of the determinands of interest has some advantages in the availability of a range of CRMs. If a nitric acid digestion method is adopted, care must be taken to select a CRM which has itself been prepared from fine-grained material.

WGMS also recommended that maps of the composition of the fine-grained material be supported by maps showing the amount of finely grained material at each sampling location.

2.5 Organic Contaminants—CBs and PAHs

Whole sediment analysis

As is the case for metals, it is possible to draw maps showing the concentrations of organic contaminants in whole sediment (usually sieved at 2 mm to exclude larger material). The resulting distribution patterns closely reflect the distribution of fine-grained sediments and organic matter, and therefore suffer from the same limitations as the comparable maps for metals.

CBs and, to a lesser degree, PAHs are often closely associated with organic matter through sorption processes. Normalization to organic carbon can therefore be applied, and maps drawn of the ratios of the concentrations of contaminants to organic carbon. The resulting maps have been used to identify spatial trends, etc., in contaminant concentrations. The mobility and partitioning of organic contaminants in the environment have been modelled on a system of partition coefficients involving water, organic matter in sediment, dissolved organic matter, organic matter in biota, etc. The ratio of the contaminant to organic carbon may therefore give a helpful indication of the availability of the contaminant to other environmental compartments, including biota.

Experience has shown that the intercepts of regressions between contaminant concentrations and organic carbon levels are commonly close to zero, i.e., normalization of organic contaminants to organic carbon does not suffer so severely from the problem of non-zero intercepts as in the case for metals. However, in sandy sediments the concentrations of the contaminants and organic carbon may be low, and subject to significant uncertainty that will be reflected in the calculated ratios. This problem is less significant for PAHs than for CBs. Improved analytical performance can reduce the errors in the ratios. WGMS recognized that it is possible to alter the analytical intake mass for organic analysis over a much wider range than that for metals, to some extent compensating for the lower concentrations in sandy sediment.

The range of concentrations of organic contaminants found in the environment is large, perhaps covering two orders of magnitude or more. It is larger than that found for metals. This tends to reduce the demands made of the normalization procedure, and may reduce the significance of non-zero intercepts, as the intercept may often be a smaller proportion of the measured concentration.

Normalization to organic carbon, and expression of the result as the equivalent concentration in sediment of a "standard" carbon content (e.g., 5 %), is not significantly different from normalization to organic carbon, as discussed above. As above, it relies upon the intercept of the regression being close to zero, and experiences increasing errors in sandy sediments.

Normalization to grain size, and expression of the result as the equivalent concentration in sediment of a standard grain size (e.g., 25 % fines), is not commonly done in relation to organic contaminants. It also relies upon the intercept of the regression of contaminant concentration on grain size being close to zero, and would experience increasing errors in sandy sediments. It is unlikely to provide more useful information than normalization to organic matter.

Procedures based on regression analysis

Procedures based on regressions between organic carbon content and contaminant concentrations in whole sediment could be employed. As in the case of metals, maps could be drawn based on residuals of data points around the regression line.

As for metals, the procedure can be improved if residuals were expressed in relation to background values. For the synthetic CBs, the background concentrations should be zero, and therefore the residuals would be equal to the measured concentrations. The procedure therefore does not yield any additional information for CBs.

The background concentrations of PAHs would not be zero, as natural processes such as forest fires and petroleum seeps contribute PAHs to the environment. Post-depositional processes such as degradation and mobilization may alter the concentrations and nature of PAHs in old sediments. WGMS felt that it would not be straightforward to establish background values for PAHs, and that use would have to be made of ecotoxicological quality objectives (EQOs) or Reference Values for comparative purposes. The resulting residuals would therefore not reflect the "degree of contamination", but would reflect the relationship between the measured value and a selected quality criterion.

Sieving and analysis of a fine fraction

WGMS agreed that sieving and the analysis of a fine fraction offered the best opportunity for a consistent and comprehensive data set describing the distribution of organic contaminants in marine sediments over wide areas. Sieving removes much of the variance arising from differences in grain size and from inhomogeneities of the organic material in the sediment (e.g., coarser material may contain plant fragments of markedly different sorptive capacity from the organic material associated with the clays). As in the case of metals, normalization would also be necessary to allow for variations in the organic carbon content of the sieved fraction.

The main advantage to be gained by sieving is that the quality of the analytical data is not degraded in coarse sediments. The separation of the fine fraction should ensure that concentrations do not approach limits of quantification of the analytical methods. The error structure of the normalized data should therefore be much simpler than that obtained from whole sediment analysis. The requirement to obtain sufficient fine material for analysis can be a significant problem, particularly in sandy sediments.

There are theoretical reasons to favour the analysis of a fine-grained fraction (such as material < 20 μm) rather than a coarser fine fraction (such as < 63 μm). However, the separation of this fraction is considerably more laborious than separation of the < 63 μm fraction, and the degradation in quality of data that would result from selection of the coarser fine fraction would probably be outweighed by the increased analytical output and reduced potential for contamination. In either case, the processing of large numbers of samples, particularly samples containing low amounts of fine material, would be greatly facilitated by the use of an automatic recirculating sieving machine.

Additional issue

Recent work on the distribution of PAHs in sediment has indicated that in some environments the presence of small quantities of soot particles can have a marked influence on the concentrations and behaviour of PAHs.

The soot particles can contain high concentrations of PAHs, and the partition coefficient between the soot and the aqueous phase can be orders of magnitude greater than for “natural” organic matter in sediment. The overall significance of soot to the environmental geochemistry of PAHs in the ICES/OSPAR areas has not been fully established. The normalization procedures recommended above do not take the presence and influence of soot into account. In interpreting data normalized to organic carbon, it is necessary to be aware of the potential role of soot.

3 ACKNOWLEDGEMENT

This work was conducted by the Working Group on Marine Sediments in Relation to Pollution (WGMS) at its 2000 meeting.

4 REFERENCES

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ANNEX 2

DETAILED METHODS FOR TEMPORAL TREND DETECTION

PART 1 ADJUSTMENT OF INPUTS

1.1 Introduction

Political targets on a substantial reduction of the quantity of nutrients and toxic substances reaching the aquatic environment of the North Sea also require the means to determine whether reduction measures taken at point and diffuse sources have been effective in reducing the inputs into the North sea. This can be done on the atmospheric inputs (as deposition) and waterborne inputs (as riverine loads). However, the input data are highly dependent on conditions, specifically on climatic influences (precipitation, flow rate, temperature). In order to prevent that climatic influences cause deterioration in trend detectability, an appropriate adjustment of annual and monthly input data prior to the analysis of temporal trends is necessary.

According to the concept adopted by the 1999 meeting of the OSPAR Working Group on Inputs to the Marine Environment (INPUT), the steps of statistical trend analysis are as described in Figure A2.1.1.1. This concept allows use of the procedure for trend analysis based on annual data described in Section 6.4 of the 1999 ACME report (ICES, 2000) also for aggregated adjusted loads.

Figure A2.1.1.1. Steps of statistical trend analysis on riverine and atmospheric inputs.

1 Provision of raw data

2 Adjustment of input load

3 Aggregation of the adjusted data

4 Trend detection and trend estimation of annual data

5 Power analysis

The aim of adjustment is to compensate for the effect of varying runoff or precipitation. Compensation may be accomplished additively or multiplicatively, based on dynamic or non-dynamic estimation methods, but multiplicative adjustment based on a dynamic estimation method is preferred (ICES, 2000).

The general procedure of multiplicative adjustment is as follows: Let c_{ij} denote the measured concentration of the jth sample in year i, taken at time t_{ij}. Let q_{ij} denote the actual runoff at that sampling time t_{ij} and q_{0ij} the corresponding long-term average runoff. Assume that there is a CQ function $c_{ij}(q_{ij})$ (concentration runoff function) describing the influence of current runoff and possibly further influence variables, so that the measured concentration can be well described by the statistical model

$$\mathrm{c}_{ij} = c_{ij}(q_{ij}) + \varepsilon_{ij},$$

where ε_{ij} denotes the respective random deviation caused by measurement error and other natural and anthropogenic influences.

The CQ function may also be used to estimate the "measured load"

$$L_{ij} = c_{ij}\, q_{ij}$$

by the "estimated load"

$$L_{ije} = c_{ij}(q_{ij})\, q_{ij},$$

or—in case the current runoff q_{ij} is not available—by the "mean load"

$$L_{ijm} = c_{ij}(q_{0ij})\, q_{0ij}.$$

Now the "adjusted load" can be calculated

$$L_{ija} = \frac{L_{ijm}}{L_{ije}} L_{ij} = \frac{c_{ij}}{c_{ij}\left(q_{ij}\right)} L_{ijm}\ ,$$

i.e., the adjusted load can be derived from the measured load by multiplying with the correction factor $\frac{L_{ijm}}{L_{ije}}$ or from the mean load by multiplying with the correction factor $\frac{c_{ij}}{c_{ij}\left(q_{ij}\right)}$.

In the following section, the concept of adjustment will be investigated in detail, focusing on its interpretation. In Section 1.3, the effect of lagged runoff is considered, and Section 1.4 describes the calculation of the power function. The statistical estimation methods are presented in Section 1.5 and the results are described in Section 1.6.

The material focuses on the adjustment of riverine loads, but the concept and the statistical methods can also be applied to atmospheric deposition.

1.2 Interpretation of Adjusted Inputs

The measured input loads are composed of many different contributions, which can be seen as caused by a set of more or less fixed sources, such as households, industry, farming, area geology, etc. Adjustment of inputs according to natural variations has to take into account the different nature of these sources. If, e.g., 80 % of the load is not affected by natural variations since it is caused by point sources, an adjustment is sensible only for the remaining 20 % of the load. For special cases, the adjusted input can be calculated as indicated in the text below.

Adjustment of loads not affected by climatic variation

Assume that there are only point sources, i.e., assume that inputs are not affected by climatic variation. Then no adjustment is required, i.e., the adjusted load L_{ija} should be equal to the actual load, $L_{ija} = L_{ij}$.

Adjustment of loads with constant concentrations

Assume that there are only diffuse sources, causing constant concentrations not depending on the runoff. Then the load L_{ij} is proportional to the runoff and it can be adjusted by multiplying with q_{0ij} / q_{ij}, i.e.,

$L_{ija} = L_{ij}\, q_{0ij} / q_{ij}$.

Adjustment of a mixture of loads from two sources

Assume that 30 %, say, of the load is caused by point sources and 70 % by loads with constant concentrations, then the aggregated adjusted load should be the sum of the single adjusted loads, i.e.,

$L_{ija} = 0.3\, L_{ij} + 0.7\, L_{ij}\, q_{0ij} / q_{ij} = (0.3 + 0.7\, q_{0ij} / q_{ij})\, L_{ij}$

If the percentages are known, no further calculation is needed. Otherwise, the question arises concerning how to calculate the percentage of the load due to diffuse sources and the percentage due to point sources.

If the CQ function is known, the percentages can be calculated as follows: Under the conditions prescribed, the CQ function can be written

$$c_{ij}(q) = \frac{\alpha_{ij}}{q} + \beta_{ij}$$

and therefore the LQ function (load runoff function) equals

$$c_{ij}(q)q = f_{ij}(q) = \alpha_{ij} + \beta_{ij} q$$

where α_{ij} denotes the load due to point sources and $\beta_{ij} q$ denotes the load due to diffuse sources. Hence, the percentage of the load due to diffuse sources can be calculated

$$\frac{\beta_{ij} q}{\alpha_{ij} + \beta_{ij} q}$$

and the percentage of the load due to point sources can be calculated

$$\frac{\alpha_{ij}}{\alpha_{ij} + \beta_{ij} q}.$$

According to this approach, the adjusted load can formally be written

$$\begin{aligned} L_{ija} &= \frac{\alpha_{ij}}{\alpha_{ij} + \beta_{ij} q_{ij}} L_{ij} + \frac{\beta_{ij} q_{ij}}{\alpha_{ij} + \beta_{ij} q_{ij}} L_{ij} \frac{q_{0j}}{q_{ij}} \\ &= \frac{\alpha_{ij} + \beta_{ij} q_{0j}}{\alpha_{ij} + \beta_{ij} q_{ij}} L_{ij} \\ &= \frac{L_{ijm}}{L_{ije}} L_{ij} \end{aligned}$$

This formula means that the adjusted load simply can be calculated by multiplying the actual load by the ratio of the estimated load at mean runoff and the estimated load at actual runoff and, in the case that 70 % of the load is due to diffuse sources, this multiplication factor equals $(0.3 + 0.7\, q_{0j} / q_{ij})$.

Note that this is equivalent with the general definition of the multiplicatively adjusted load given in Section 1.1, above.

Adjustment of loads due to a non-linear load-runoff function

Assume that the load follows the LQ function

$$L_{ij} = f_{ij}(q_{ij}) = \gamma_{ij} q_{ij}^d .$$

Adjustment means that the actual runoff q_{ij} will be replaced by the long-term mean runoff q_{0ij}. Since $\gamma_{ij} = \frac{L_{ij}}{q_{ij}^d}$, this leads to

$$L_{ija} = \gamma_{ij} q_{0j}^d = \frac{q_{0j}^d}{q_{ij}^d} L_{ij} = \frac{f(q_{0j}^d)}{f(q_{ij}^d)} L_{ij} .$$

This means again that the adjusted load simply can be calculated by multiplying the actual load by the ratio of the estimated load at mean runoff and the estimated load at actual runoff.

Adjustment of loads due to a combination of three sources

Assume, for example, that 30 % of the load is caused by point sources, 50 % by sources causing constant concentrations, and 20 % by sources causing a non-linear load-runoff function, then the aggregated adjusted load should be the sum of the single adjusted loads, i.e.,

$$L_{ija} = 0.3\, L_{ij} + 0.5\, L_{ij}\, q_{0j} / q_{ij} + 0.2\, L_{ij}\, (q_{0ij} / q_{ij})^d.$$

If the percentages are known, no further calculation is needed. Otherwise, the question arises as to how to calculate these percentages. As in the case of adjustment of a mixture of loads from two sources, if the CQ function is known, the percentages can be derived from the CQ or the LQ function, respectively. Under the conditions prescribed, the LQ function can be written

$$c_{ij}(q)q = \alpha_{ij} + \beta_{ij} q + \gamma_{ij} q^d$$

where α_{ij} denotes the load due to point sources, $\beta_{ij}q$ denotes the load due to diffuse sources (linear), and $\gamma_{ij}q^d$ denotes the load due to the non-linear component. Then the percentages of the load due to point sources can be calculated

$$\frac{\alpha_{ij}}{\alpha_{ij} + \beta_{ij} q + \gamma_{ij} q^d},$$

the percentage of the load due to diffuse sources can be calculated

$$\frac{\beta_{ij} q}{\alpha_{ij} + \beta_{ij} q + \gamma_{ij} q^d},$$

and the percentage of the load due to the non-linear component can be calculated

$$\frac{\gamma_{ij} q^d}{\alpha_{ij} + \beta_{ij} q + \gamma_{ij} q^d}.$$

According to this approach, the adjusted load can formally be written

$$\begin{aligned} L_{ija} &= \frac{\alpha_{ij}}{\alpha_{ij} + \beta_{ij} q_{ij} + \gamma_{ij} q_{ij}^d} L_{ij} + \frac{\beta_{ij} q_{ij}}{\alpha_{ij} + \beta_{ij} q_{ij} + \gamma_{ij} q_{ij}^d} L_{ij} \frac{q_{0ij}}{q_{ij}} \\ &+ \frac{\gamma_{ij} q_{ij}^d}{\alpha_{ij} + \beta_{ij} q_{ij} + \gamma_{ij} q_{ij}^d} L_{ij} \left(\frac{q_{0ij}}{q_{ij}} \right)^d \\ &= \frac{\alpha_{ij} + \beta_{ij} q_{0ij} + \gamma_{ij} q_{0ij}^d}{\alpha_{ij} + \beta_{ij} q_{ij} + \gamma_{ij} q_{ij}^d} L_{ij} = \frac{L_{ijm}}{L_{ije}} L_{ij}. \end{aligned}$$

Again, this is in accordance with the definition of the adjusted load in Section 1.1. Hence, it can be concluded that the general concept of multiplicative adjustment is in accordance with the interpretation for special cases given earlier in this section.

1.3 Effects of Lagged Runoff

The adjustment of inputs aims at reducing the interannual variability of loads. However, adjustment is performed at the level of single measurements, whereas the interannual variability is based on aggregated values. As long as the CQ function is not dependent on lagged runoff, this should not cause any problem. In order to obtain a better understanding of possible lag effects, a simulation study applying a very simple simulation model was performed. It is assumed that daily measurements are available, and that the daily runoff is Normally distributed according to the model

$$q_{ij} = 6 + u_i + v_{ij},$$

where u_i and v_{ij} denote random variables reflecting variations between years and within years. The measured load follows the model

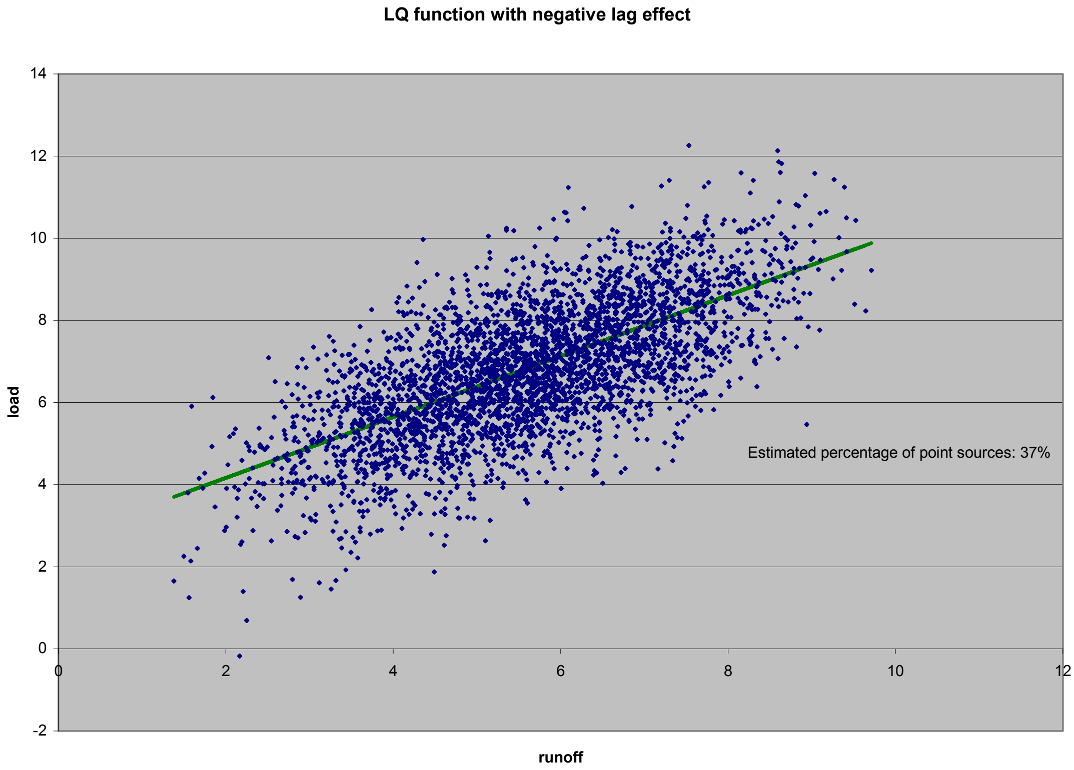
$$L_{ij} = 4 + q_{ij} - p \times q_{i,j-1} + w_{ij},$$

where p denotes the lag effect of the runoff of the day before. All random variables u_i, v_{ij} und w_{ij} are stochastically independent and standard Normally distributed. Note that the mean load is constant over the years.

The simulation study was performed for ten years and several ps. In each case, the model parameters were calculated by using a simple regression approach, and then the adjusted loads were calculated according to the formulas given in Section 1.1. In case that there is no lag effect ($p = 0$), this approach leads to substantial reduction of the interannual variability.

But for very large lag effects, this method may fail. Figure A2.1.3.1 shows the resulting 3650 pairs of load and runoff obtained in the simulation study for $p = -0.5$.

Figure A2.1.3.1. Results of a simulation study for ten years and $p = -0.5$ (see text for explanation).



The straight line represents the linear LQ function obtained from simple linear regression, into account the negative lag effect. The resulting annual adjusted load is presented in Figure A2.1.3.2 (type I). It is clearly "overadjusted". This can be explained by the fact that the percentage of diffuse sources is overestimated when the negative lag effect is ignored. A better result will be obtained if the lag effect would be taken into account in the regression model. The resulting adjusted load (type II) is almost constant over the years.

It can be concluded that lag effects of the runoff may lead to over-adjustment (in case of negative lag effects) as well as to under-adjustment (in case of positive lag effects), and in both cases the gain of adjustment may be deteriorated substantially. Therefore, in case of non-satisfying reduction of the interannual variability, it is strongly recommended to check possible lag effects by calculating the correlation function between the residuals c_{ij}–c_{ij} (q_{ij}) and the lagged runoff $q_{i,-r}$ for $r = 1,2,3,\ldots$ If there is a significant correlation, the statistical model should be extended in a suitable manner.

1.4 Estimation Methods

According to the concept described in Section 1.1, a statistical model for load and concentration is needed in order to estimate concentration and load as functions of runoff. Figure A2.1.4.1 contains the load-runoff diagram for nitrate measured at Lobith/Rhine biweekly from 1955 to 1995. Apparently there is an approximately linear relation between load and runoff.

Figure A2.1.4.2 contains the results for Total P. Again, the relation between runoff and load could be modelled linearly, but there is more variability and heterogeneity of variance.

For other rivers and other nutrient parameters, similar diagrams can be obtained. For heavy metals, the relation between runoff and load is frequently less clear. Figure A2.1.4.3 shows the load-runoff diagram for cadmium.

A linear LQ function is equivalent to the assumption that the CQ function may be modelled as a linear function of the reciprocal runoff. This is the basic assumption of the methods N, H, L1, L2, L3, and L4 which are described below.

Method N: Smoothing with non-parametric constraints

Modifying an approach considered in the HARP Guidelines 7, the following dynamic model is considered:

$$L_{ij} = \alpha_{ij} + \beta_{ij} q_{ij} + \varepsilon_{ij}$$

with

L_{ij} = load in year i and season j

q_{ij} = runoff in year i and season j

$i = 1,\ldots,T$

$j = 1,\ldots,M.$

Figure A2.1.3.2. Annual loads, with and without taking the negative lag effect into account.

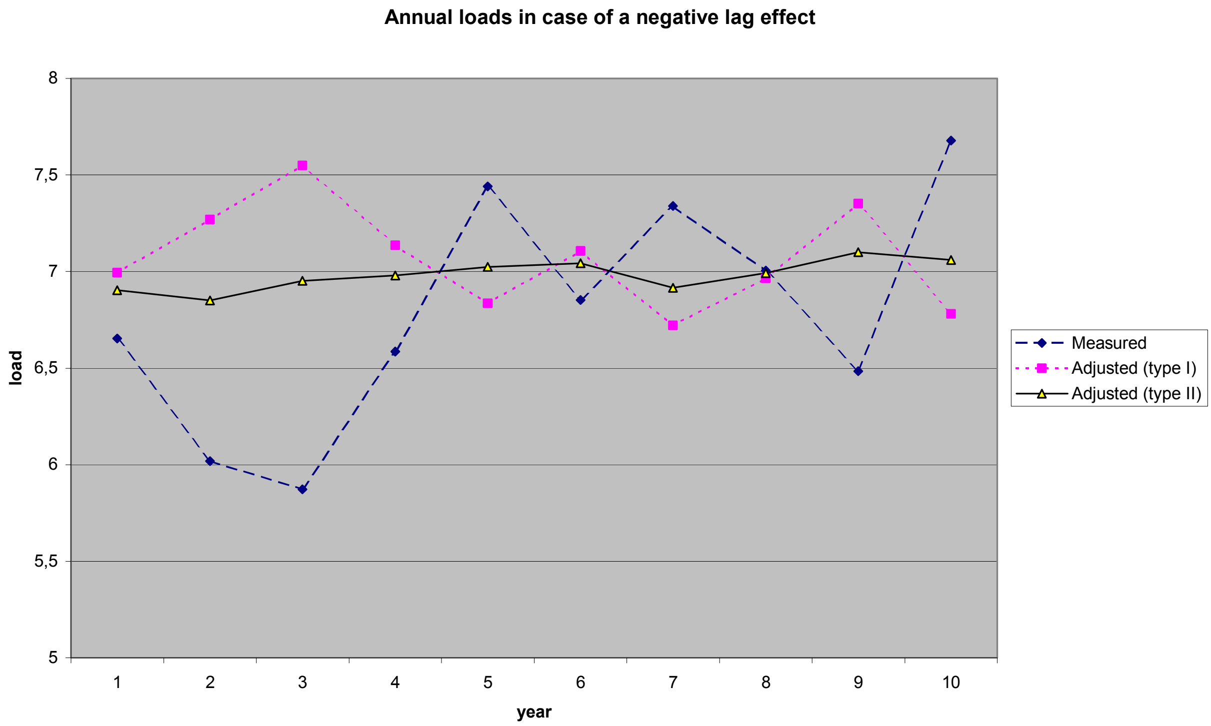


Figure A2.1.4.1. Load-runoff diagram for nitrate (g s^{-1}) measured biweekly at Lobith/Rhine from 1955–1995.

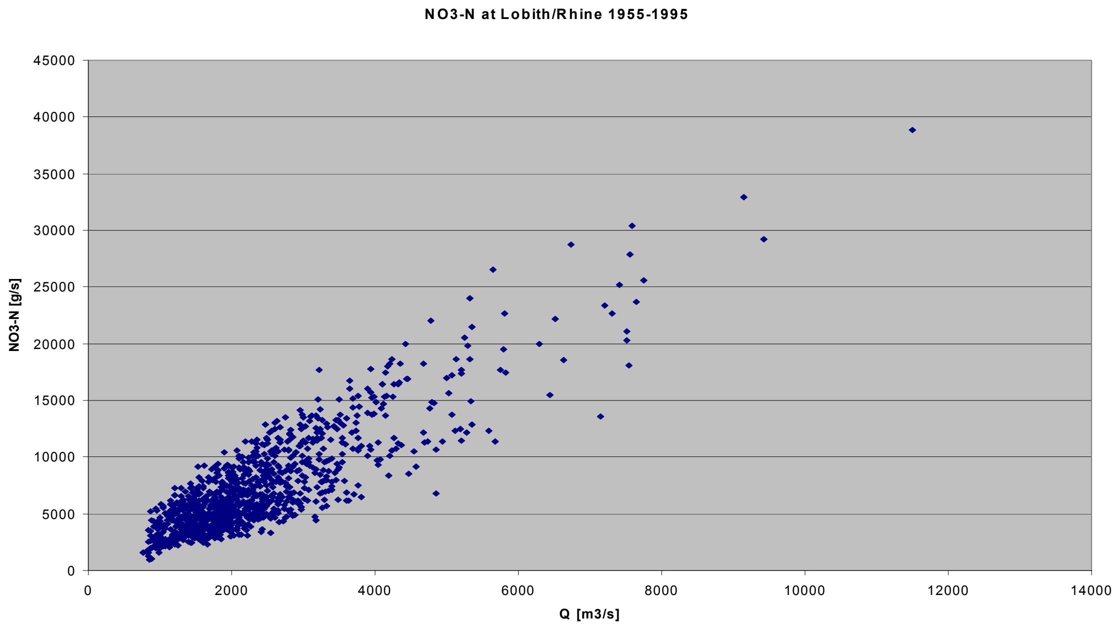


Figure A2.1.4.2. Load-runoff diagram for Total P (g s^{-1}) measured at Lobith/Rhine from 1973–1995.

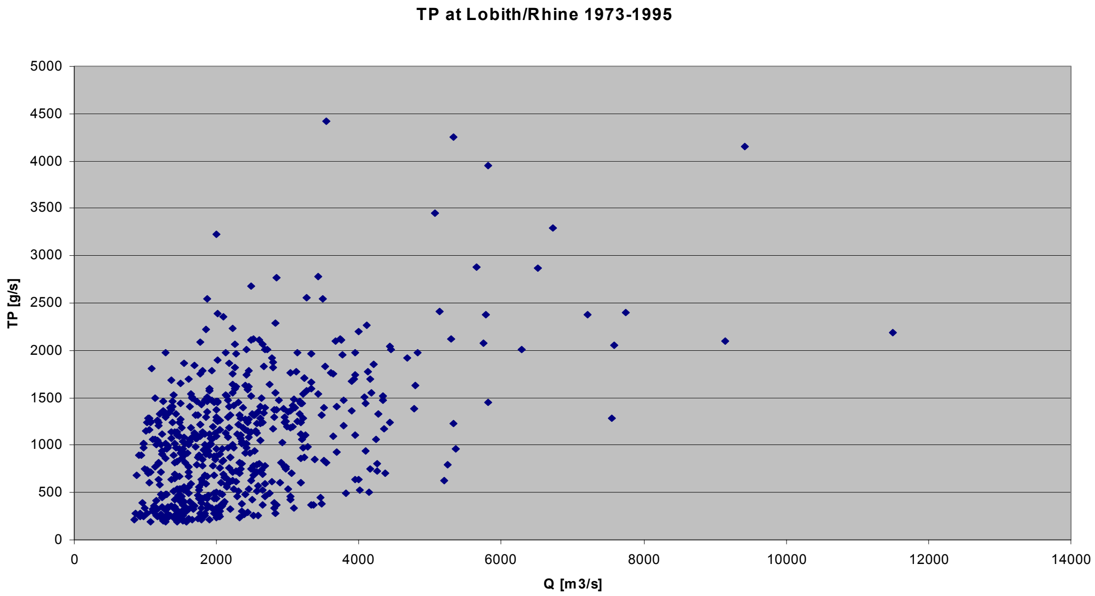
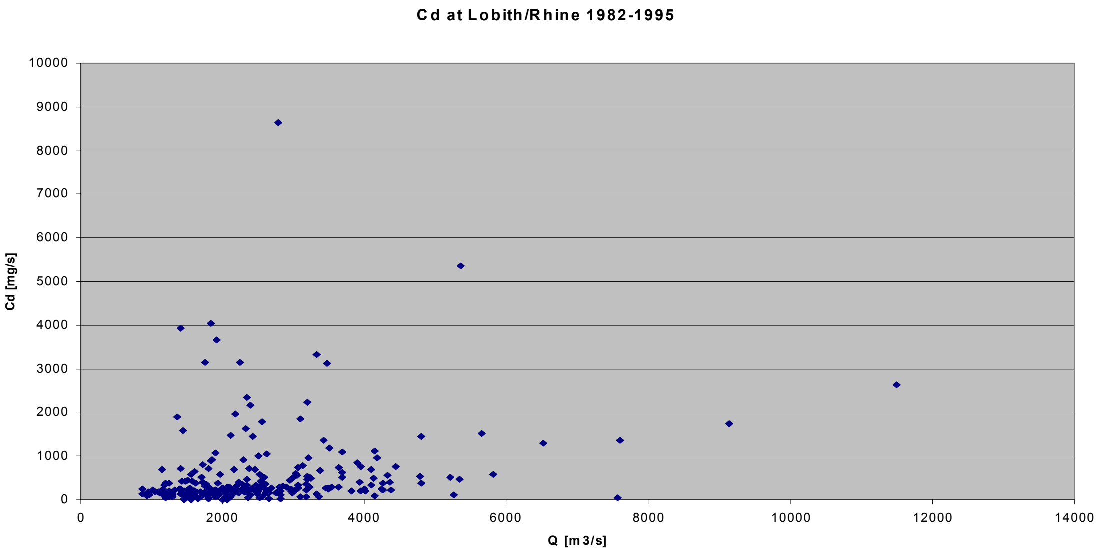


Figure A2.1.4.3. Load-runoff diagram for cadmium (mg s^{-1}) measured at Lobith/Rhine from 1982–1995.



According to this model, the multiplicatively adjusted load can be calculated

$$L_{ija} = \frac{\alpha_{ij} + \beta_{ij} q_{0j}}{\alpha_{ij} + \beta_{ij} q_{ij}} L_{ij} = \frac{\dfrac{\alpha_{ij}}{\beta_{ij}} + q_{0j}}{\dfrac{\alpha_{ij}}{\beta_{ij}} + q_{ij}} L_{ij} \,,$$

i.e., the adjustment step only depends on the ratio α_{ij} / β_{ij}.

The model parameters α_{ij} and β_{ij} are estimated by minimizing an expression of the form:

$$\begin{aligned} S(\alpha,\beta) &= \sum_{i,j} (L_{ij} - \alpha_{ij} - \beta_{ij} q_{ij})^2 \\ &+ \lambda_1 \sum_{i,j} (\alpha_{ij} - \frac{\alpha_{i+1,j} + \alpha_{i-1,j}}{2})^2 + \lambda_2 \sum_{i,j} (\alpha_{ij} - \frac{\alpha_{i,j+1} + \alpha_{i,j-1}}{2})^2, \\ &+ \lambda_1 \overline{q^2} \sum_{i,j} (\beta_{ij} - \frac{\beta_{i+1,j} + \beta_{i-1,j}}{2})^2 + \lambda_2 \overline{q^2} \sum_{i,j} (\beta_{ij} - \frac{\beta_{i,j+1} + \beta_{i,j-1}}{2})^2 \end{aligned}$$

with $\overline{q^2} = \frac{1}{N} \sum_{ij} q_{ij}^2$ and penalty factors λ_1, λ_2 to define a desired compromise between overfitting and specification errors. Suitable levels of the penalty factors can be established by undertaking a cross-validation study of relationships between L_{ij} and q_{ij}. In the examples presented here, further restrictions are posed: the generalized degrees of freedom of the model are constant and $\lambda_1 = \lambda_2$.

Method H: Approach of Hebbel

According to a general approach of Hebbel (1992), the following decomposition of the concentration series into trend, season, and exogenous effects may be considered:

$$c(t) = u(t) + s(t) + \beta / q(t) + \varepsilon(t)$$

or equivalently

$$L(t) = q(t) \times (u(t) + s(t) + \beta / q(t) + \varepsilon(t))$$

where

$c(t)$ = concentration at time t
$L(t)$ = load at time t
$q(t)$ = runoff or precipitation at time t
$u(t)$ = trend,
$s(t)$ = season,
β = effect of runoff/precipitation in the concentration.

According to this model, the multiplicatively adjusted load can be written

$$L_a(t) = \frac{\beta + (u(t)+s(t))q_0(t)}{\beta + (u(t)+s(t))q(t)} L(t) = \frac{\dfrac{\beta}{u(t)+s(t)} + q_0(t)}{\dfrac{\beta}{u(t)+s(t)} + q(t)} L(t),$$

where $q_0(t)$ denotes the long-term mean in the month corresponding to t.

The sampling times $t_1,...,t_n$ are not necessarily equidistant. In order to describe the estimation procedure, matrix notation will be used. Let $c = (c(t_1),...,c(t_n))'$ denote the concentration vector, $u = (u(t_1),...,u(t_n))'$ and $s = (s(t_1),...,s(t_n))'$ the vectors of trend and season, and $A = (1/q(t_1),...,1/q(t_n))'$ the design matrix belonging to β. Under the assumption that β is known, trend u and season s can be estimated with an appropriate linear smoother according to the equation

$$\hat{u} + \hat{s} = S(c - A\beta), \qquad [1]$$

where S denotes the smoother matrix. Under the assumption that $u + s$ is known, β can be estimated by linear regression according to the equation

$$\hat{\beta} = (A'A)^{-1} A'(c - (u+s)). \qquad [2]$$

Letting the estimates equal to the estimated parameters, the equations [1, 2] are equivalent to

$$\hat{u} + \hat{s} = S(c - A\hat{\beta})$$

with

$$\hat{\beta} = (A'(I-W)A))^{-1} A'(I-W)c).$$

Here, I denotes the identity matrix. It should be noted that in the model applied, β is a constant. However, due to reduction of inputs β may change over time, and therefore it is recommended to apply the procedure locally for a time window of given length. In the calculations presented, a seven-year window was applied.

The estimation method works with every linear smoother. In the calculations presented, the method described by Hebbel (1997) is applied. According to this method, trend and seasonality are estimated by minimizing a roughness functional. This functional depends on the polynomial order for the trend, on the order of seasonality and the smoothness parameter σ^2. In the calculations presented, a linear trend and seasonality of second order were used. The smoothness parameter is determined so that the generalized degrees of freedom are about 12 per seven years.

Method L1: Local regression with seasonality

If in method H the smoothness parameter σ^2 is tending to infinity, the estimation method becomes equivalent to a linear regression analysis based on the following model:

$$c(t) = \frac{\alpha}{q(t)} + \beta + \delta t + \gamma_1 \sin\frac{2\pi t}{m} + \gamma_2 \cos\frac{2\pi t}{m} + \gamma_3 \sin\frac{2\pi t}{2m} + \gamma_4 \cos\frac{2\pi t}{2m} + \varepsilon(t)$$

Here m denotes the length of the year. According to the notation of method H, the trend is described by

trend = $\beta + \delta t$

and the season is described by

season =

$$\gamma_1 \sin\frac{2\pi t}{m} + \gamma_2 \cos\frac{2\pi t}{m} + \gamma_3 \sin\frac{2\pi t}{2m} + \gamma_4 \cos\frac{2\pi t}{2m}.$$

The estimation procedure is applied locally for a time window of seven years, i.e., for every sampling time t the parameters α, β, δ, and γ_1, γ_2, γ_3, γ_4 are locally estimated with all data of the corresponding time window.

With this model, the multiplicatively adjusted load can be written

$$L_a(t) = \frac{\alpha + \left(\beta + \delta t + season\right) q_0(t)}{\alpha + \left(\beta + \delta t + season\right) q(t)} L(t)\ .$$

Method L2: Local regression with seasonality and lagged runoff effect

If the load is not only depending on the current runoff but also on lagged runoffs, the model used for method L1 can be extended as follows:

$$c(t) = \frac{\alpha}{q(t)} + \frac{\gamma}{q(t-1)} + \beta + \delta t + \gamma_1 \sin\frac{2\pi t}{m} + \gamma_2 \cos\frac{2\pi t}{m} + \gamma_3 \sin\frac{2\pi t}{2m} + \gamma_4 \cos\frac{2\pi t}{2m} + \varepsilon(t)$$

where $q(t-1)$ denotes the runoff of the day before (which is frequently available). With this model, the multiplicatively adjusted load can be written

$$L_a(t) = \frac{\alpha + \gamma + \left(\beta + \delta t + season\right) q_0(t)}{\alpha + \gamma \dfrac{q(t)}{q(t-1)} + \left(\beta + \delta t + season\right) q(t)} L(t)$$

where $q_0(t)$ denotes the long-term mean in the month corresponding to t. The parameters are estimated using local regression. In order to improve the stability of the estimated parameters, a time window of eight instead of seven years is used.

Method L3: Local regression with lagged runoff effect and water temperature

In order to simplify the CQ function, one idea is to replace the second order seasonal component (four unknown parameters) by a temperature effect which requires only one unknown parameter (η):

$$c(t) = \frac{\alpha}{q(t)} + \frac{\gamma}{q(t-1)} + \beta + \delta t + \eta w(t) + \varepsilon(t)$$

where $w(t)$ denotes the water temperature on day t. Estimation of the unknown parameters is performed with local linear regression with an eight-year time window.

Method L4: Local regression with seasonality, lagged runoff effect and water temperature

Combining the models applied for the methods L2 and L3 results in the following CQ model:

$$c(t) = \frac{\alpha}{q(t)} + \frac{\gamma}{q(t-1)} + \beta + \delta t + \eta w(t) + \gamma_1 \sin\frac{2\pi t}{m} + \gamma_2 \cos\frac{2\pi t}{m} + \gamma_3 \sin\frac{2\pi t}{2m} + \gamma_4 \cos\frac{2\pi t}{2m} + \varepsilon(t)$$

Estimation of the unknown parameters is performed with local linear regression with an eight-year time window.

Method S1: CQ-Spline

The methods described above assume that the LQ function is approximately linear. If this is not adequate, the CQ function may be modelled with a cubic spline. If $s(.)$ denotes the spline function, the CQ function can be written

$$c_t = s(q_t) + \varepsilon_t$$

and the accordingly adjusted load equals

$$L_a(t) = \frac{s\left(q_0(t)\right) q_0(t)}{s\left(q(t)\right) q(t)} L(t)\ ,$$

i.e., the adjustment is determined by (1) the ratio of current runoff and long-term mean runoff, and (2) the ratio of the respective spline values.

The smoothing parameter of the spline function is determined so that the generalized degrees of freedom equals four.

According to method S1 the spline function is calculated on the basis of all available data, i.e., the adjustment step is not accommodated for temporal changes.

Method S2: CQ-Spline locally calculated

In order to accommodate the CQ function for temporal changes, the spline may be calculated locally with a three-year time window. This means that the spline $s(.)$ is recalculated for every sampling time t.

1.5 Results and Discussion

The methods described in the preceding section were applied to seven parameters (NO_3-N, NH_4-N, total P, *ortho*-P, Cd, Pb, and suspended matter), measured biweekly in the Rhine River (Lobith) and monthly in the Ems River (Herbrum).

The aim is to identify a method which allows efficient adjustment, i.e., an adjustment which results in annual indices not influenced by climatic variation. Therefore, the interannual variation of these indices should be smaller than for the unadjusted load. It is quantified by the power for detecting a linear trend according to the method described in Section 1.4. It should be noted that this quantification is based on the assumption that there is no serial correlation in the series of adjusted loads and that serial correlation induced by the estimation step can be neglected. Since the methods considered here are based on sparse statistical models this is assumed to hold, and this assumption is confirmed by the calculations presented in Section 6.6.1, item b) in the body of this report.

This section contains the results of the analyses for nitrate at Lobith/Rhine in detail and an overview of the estimated power of the methods for the different parameters.

Nitrate at Lobith/Rhine

Figure A2.1.5.1 shows NO_3-N concentrations (solid line) at Lobith/Rhine from 1955–1995 and the corresponding runoff series (squares).

A model fit for the concentration series based on method L1 explains 86.6 % of the variance, and the residue standard deviation is 0.35 mg l^{-1}. Figure A2.1.5.2 contains a plot of the residuals.

In order to assess the influence of lagged runoff, the correlation between the series of residuals and lagged runoff was calculated, but no clear dependency could be detected. There is some serial correlation in the series of the residuals, which is apparent in Figure A2.1.5.3. Apparently this is not a white-noise process and therefore it can be concluded that method L1 does not allow an optimal fit of the data. The fit could be improved by a time window of five instead of seven years, or by use of method H. With method H, 88.6 % of the variance can be explained, and the corresponding residue standard deviation is 0.32 mg l^{-1}. The correlogram (Figure A2.1.5.4) shows that some of the serial correlations have now disappeared.

One can conclude regarding the model fit that method H is better than method L1. However, regarding the annual adjusted loads, the differences between these methods can be neglected. The averaged relative deviation between these loads is 0.3 %, whereas the averaged deviation to the load calculated with the OSPAR formula is about 15 %. Figure A2.1.5.5 contains the annual adjusted loads according to methods N, H, L1, L4, S1 and the load according to the “OSPAR formula”. Apparently the differences between the adjusted loads are small compared to the differences to the OSPAR load. Similar results were obtained for the other adjustment methods.

Overall Results

Tables A2.1.5.1 and A2.1.5.2 contain the estimated power values for a 20 % reduction and a 50 % reduction, respectively, of inputs for seven parameters measured in the rivers Rhine and Ems. The calculation of the *post-hoc* power is based on the seven-year LOESS smoother,

Figure A2.1.5.1. Concentrations of nitrate in mg l^{-1} (solid line) at Lobith/Rhine from 1955–1995 and the corresponding runoff in $m^3\ s^{-1}$ (squares).

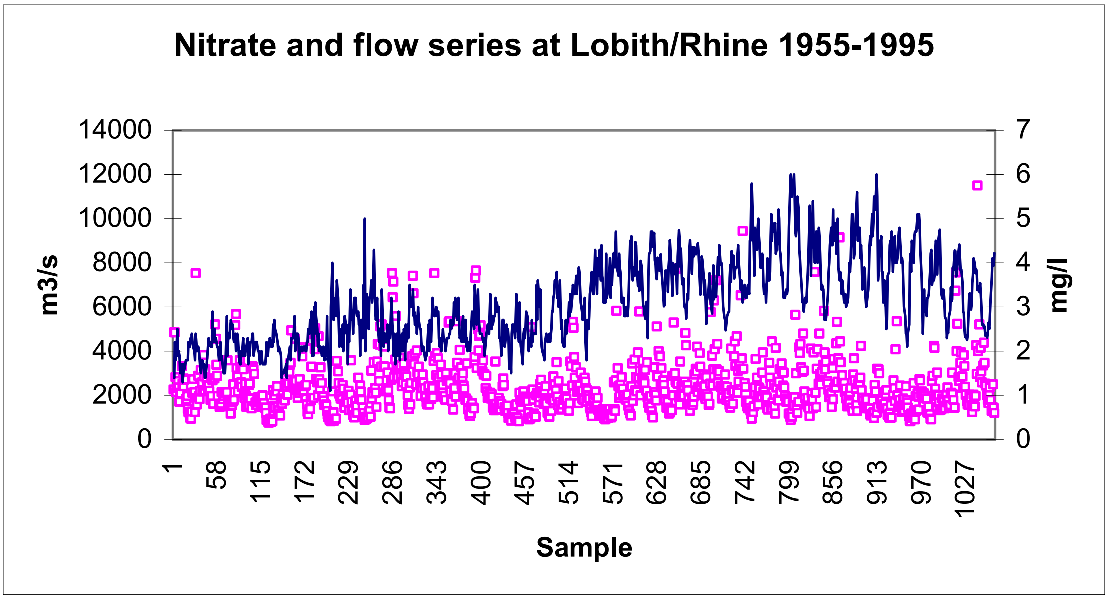


Figure A2.1.5.2. Plot of the residuals obtained for method L1 applied to the nitrate data from Lobith/Rhine (see Figure A2.1.5.1).

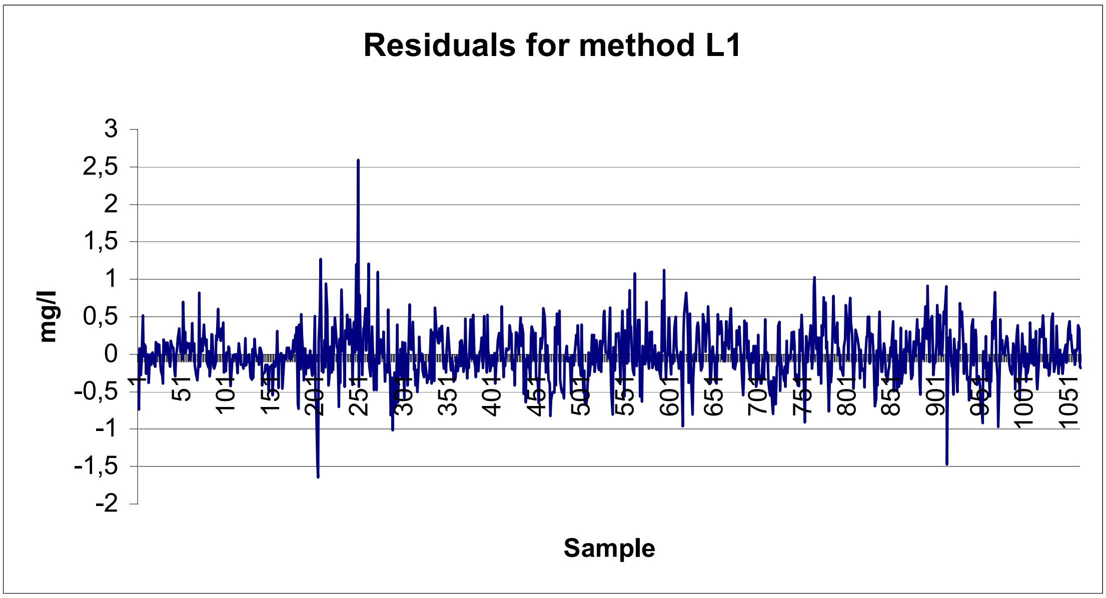


Figure A2.1.5.3. Correlogram of residuals using method L1.

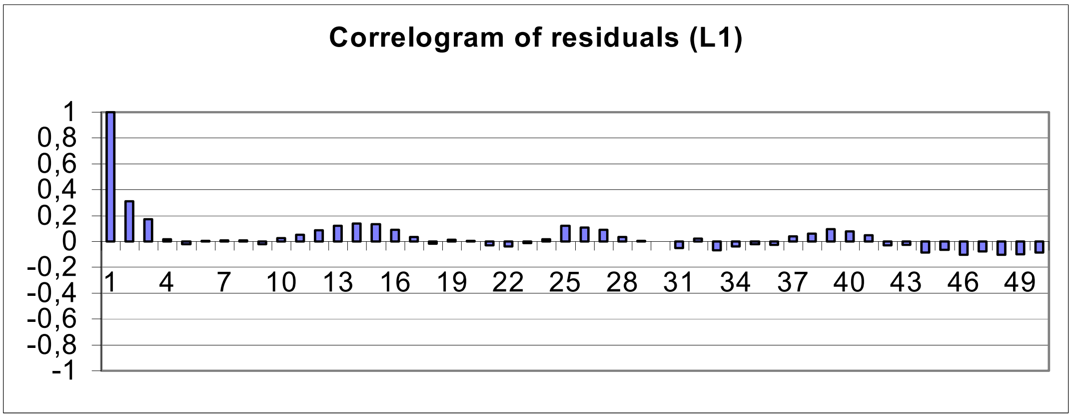


Figure A2.1.5.4. Correlogram of residuals using method H.

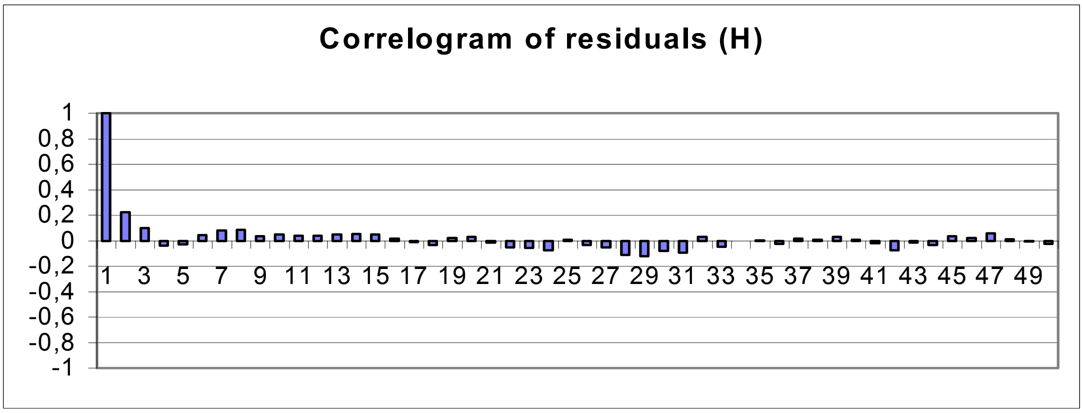
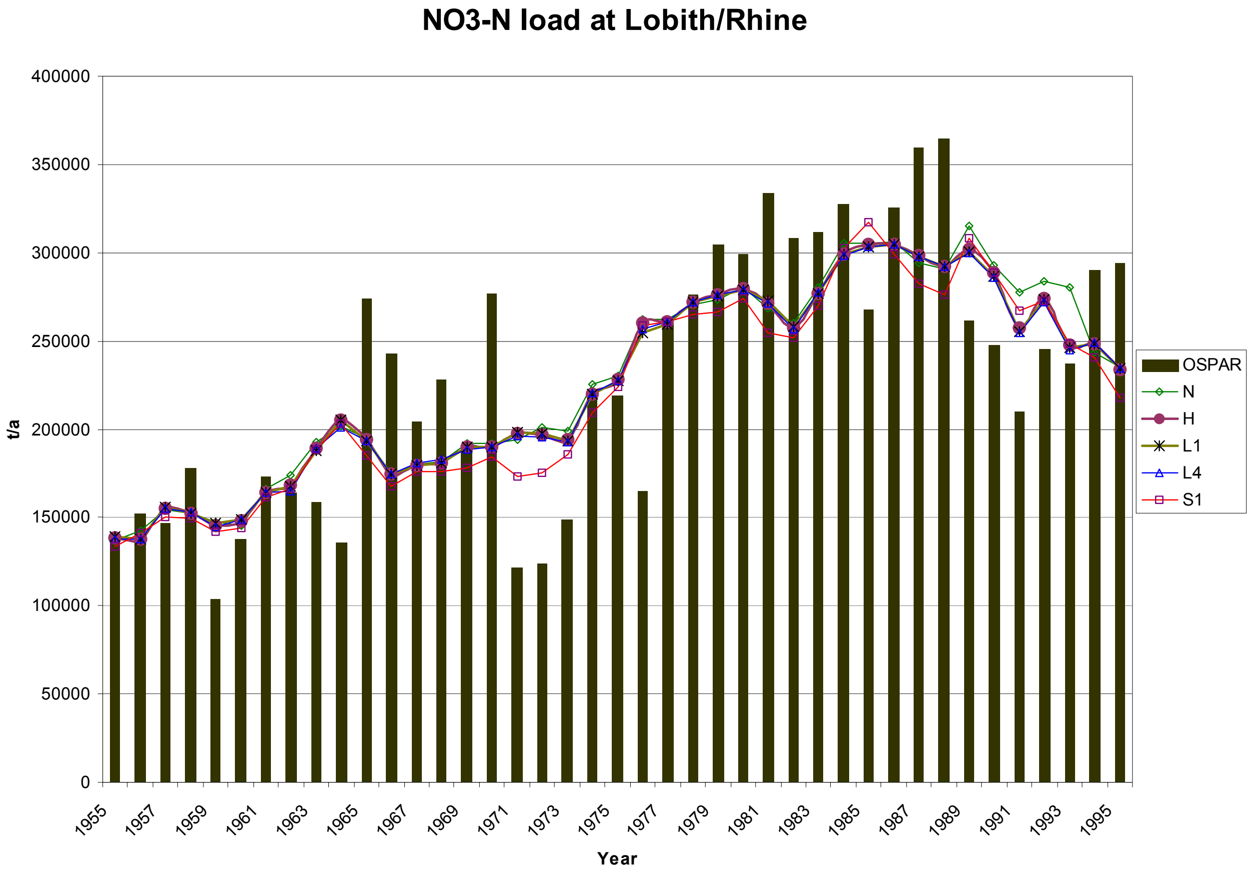


Figure A2.1.5.5. Annual adjusted loads of nitrate at Lobith/Rhine according to methods N, H, L1, L4, S1 and the load according to the OSPAR formula.



assuming that the annual loads are stochastically independent and Normally distributed and that there is a linear downward trend. Due to the large differences of the level, the calculation is based on the standard deviation of relative results. The number of years observed is $n = 10$ and the significance level is $\alpha = 0.05$. The power was calculated:

a) for all adjustment methods described before,

b) for the annual mean values of the measured concentrations, i.e.,

$$\frac{1}{M}\sum_{j=1}^{M} c_{ij} \text{ , and}$$

c) for the long-term runoff-corrected load

$$\frac{q_0}{\sum_{j=1}^{M} q_{ij}} \sum_{j=1}^{M} c_{ij} q_{ij} \qquad \text{(method A0).}$$

Here, M denotes the number of samples in year i, and q_0 denotes the long-term mean runoff.

It should be noted that the lengths of the times series are between 5 and 41 years, and for short series with less than 12 to 15 years the results presented are very crude estimates of the power.

For the Rhine River, the use of adjusted loads instead of the OSPAR load increases trend detectability considerably for nitrate, total P, ortho-P, and suspended matter, whereas for the other substances only small differences can be observed. Using concentration mean values reduces the power substantially. Only for nitrate, ortho-P, and suspended matter do concentration mean values behave better than the OSPAR load (although worse than adjusted loads). For the Ems River, the use of adjusted loads instead of the OSPAR load increases trend detectability for all nutrients. The same holds for the annual mean concentration values. The power for the concentration mean values is high since the runoff of the river Ems underlies large seasonal effects with a maximum in winter. Hence the variability of loads (OSPAR load and adjusted loads) is dominated by the variability of the measured concentrations in winter, whereas for the concentration mean these fluctuations are smoothed over the year. However, one should note that in this case of strong seasonality the annual concentration mean may not properly reflect temporal changes of inputs.

Table A2.1.5.1. Power at 20 % reduction within ten years.

Station	Parameter	Time series	Concentration	OSPAR	A0	N	H12	L1	L2	L3	L4	S1	S2
Lobith	NO_3-N	1955–1995	80 %	20 %	80 %	94 %	93 %	94 %	95 %	95 %	95 %	84 %	82 %
Lobith	NH_4-N	1955–1995	17 %	27 %	17 %	27 %	27 %	27 %	29 %	28 %	29 %	28 %	25 %
Lobith	Total-P	1973–1995	35 %	41 %	33 %	60 %	66 %	58 %	63 %	58 %	58 %	52 %	49 %
Lobith	Ortho-P	1973–1993	25 %	20 %	23 %	33 %	30 %	32 %	30 %	31 %	28 %	28 %	37 %
Lobith	Cd	1982–1995	14 %	14 %	14 %	14 %	16 %	17 %	17 %	18 %	17 %	17 %	13 %
Lobith	Pb	1982–1995	26 %	31 %	27 %	28 %	27 %	29 %	29 %	29 %	25 %	31 %	24 %
Lobith	Suspended matter	1991–1995	33 %	12 %	16 %	60 %	24 %	27 %	24 %	16 %	12 %	23 %	78 %
Herbrum	NO_3-N	1982–1997	61 %	17 %	33 %	33 %	54 %	54 %	55 %	53 %	53 %	38 %	30 %
Herbrum	NH_4-N	1982–1997	13 %	9 %	12 %	16 %	16 %	13 %	12 %	12 %	12 %	15 %	14 %
Herbrum	Total-P	1982–1997	27 %	11 %	24 %	25 %	28 %	27 %	30 %	30 %	30 %	31 %	26 %
Herbrum	Ortho-P	1984–1997	13 %	9 %	12 %	14 %	15 %	15 %	15 %	15 %	15 %	14 %	15 %
Herbrum	Cd	1993–1997	8 %	12 %	16 %	9 %	11 %	10 %	10 %	10 %	9 %	10 %	9 %
Herbrum	Pb	1993–1997	73 %	18 %	16 %	7 %	32 %	53 %	82 %	74 %	87 %	52 %	61 %
Herbrum	Suspended matter	1982–1997	22 %	13 %	14 %	21 %	21 %	20 %	21 %	20 %	20 %	21 %	18 %

Table A2.1.5.2. Power at 50 % reduction within ten years.

Station	Parameter	Time series	Concentration	OSPAR	A0	N	H12	L1	L2	L3	L4	S1	S2
Lobith	NO_3-N	1955–1995	100 %	63 %	100 %	100 %	100 %	100 %	100 %	100 %	100 %	100 %	100 %
Lobith	NH_4-N	1955–1995	53 %	82 %	54 %	82 %	82 %	81 %	85 %	84 %	86 %	83 %	78 %
Lobith	Total-P	1973–1995	93 %	97 %	91 %	100 %	100 %	100 %	100 %	100 %	100 %	99 %	99 %
Lobith	Ortho-P	1973–1993	77 %	64 %	72 %	91 %	87 %	90 %	88 %	88 %	83 %	83 %	95 %
Lobith	Cd	1982–1995	41 %	39 %	42 %	41 %	50 %	52 %	53 %	56 %	52 %	54 %	37 %
Lobith	Pb	1982–1995	81 %	89 %	82 %	84 %	86 %	85 %	85 %	86 %	77 %	88 %	75 %
Lobith	Suspended matter	1991–1995	91 %	32 %	50 %	100 %	75 %	81 %	76 %	48 %	30 %	73 %	100 %
Herbrum	NO_3-N	1982–1997	100 %	51 %	91 %	91 %	100 %	100 %	100 %	100 %	100 %	95 %	88 %
Herbrum	NH_4-N	1982–1997	38 %	18 %	31 %	49 %	49 %	35 %	34 %	33 %	33 %	45 %	39 %
Herbrum	Total-P	1982–1997	83 %	29 %	76 %	79 %	84 %	83 %	87 %	87 %	88 %	88 %	80 %
Herbrum	Ortho-P	1984–1997	38 %	20 %	33 %	39 %	45 %	45 %	46 %	44 %	45 %	39 %	46 %
Herbrum	Cd	1993–1997	13 %	34 %	49 %	19 %	27 %	25 %	22 %	21 %	17 %	21 %	21 %
Herbrum	Pb	1993–1997	100 %	58 %	48 %	14 %	90 %	100 %	100 %	100 %	100 %	99 %	100 %
Herbrum	Suspended matter	1982–1997	69 %	36 %	40 %	65 %	66 %	64 %	66 %	64 %	65 %	69 %	58 %

It turns out that there is no method that is optimal for every river and every substance. However, method L1 performs reasonably well in terms of smoothness for nutrients and partly also for heavy metals. Method L2 has slight advantages in case there is a significant effect of the lagged runoff, and methods S1 and S2 have advantages in case the LQ function is not linear. The use of the annual concentration mean cannot be recommended since there are cases where this method is very poor and where adjustment leads to much better results (e.g., for nutrients in the river Rhine).

1.6 Acknowledgement

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1.7 References

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PART 2 LINEAR TREND DETECTION IN TIME SERIES SHOWING SERIAL CORRELATION AND SEASONALITY

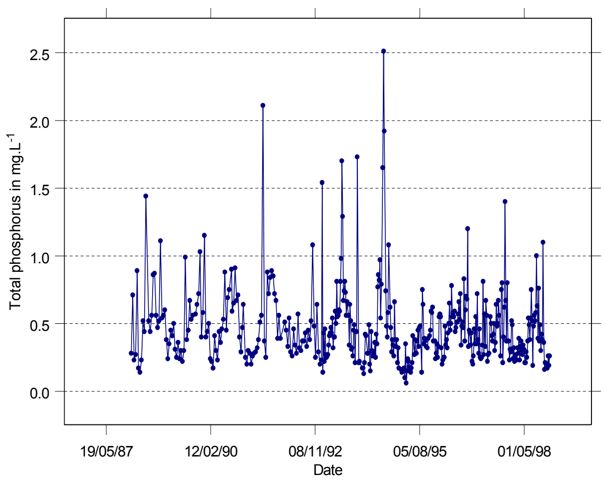
2.1 Introduction

The purpose of this section is, by use of an example, to present different approaches for linear trend detection in time series showing serial correlation and seasonality. Two different approaches are presented where season is modelled versus fitting a trend on some annual indices such as yearly means or medians. Section 2.2, below, contains a description of the data set, and the two approaches are described in Sections 2.3 and 2.4.

2.2 Data Set

To illustrate the effect of monthly measured values instead of yearly values (this could be the mean or median values of the underlying monthly data) on the trend detection method, the total phosphorus values (mg l^{-1}) measured at the location Eijsden in the Meuse River were used. This is a location near the Dutch-Belgian border. The monitoring programme started in 1988 on a biweekly frequency; in 1993 and onwards the frequency was changed to nearly every week. The time series is shown in Figure A2.2.2.1.

Figure A2.2.2.1. The time series of total phosphorus (mg l^{-1}) at Eijsden in the Meuse River (1987–1998).

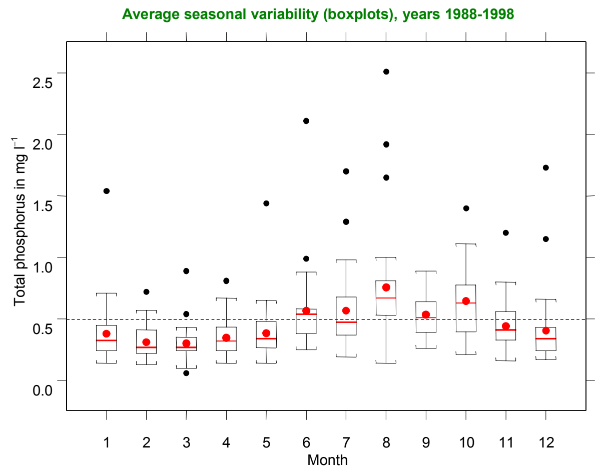


2.3 Statistical Analysis: First Approach

The statistical analysis is presented step by step, below.

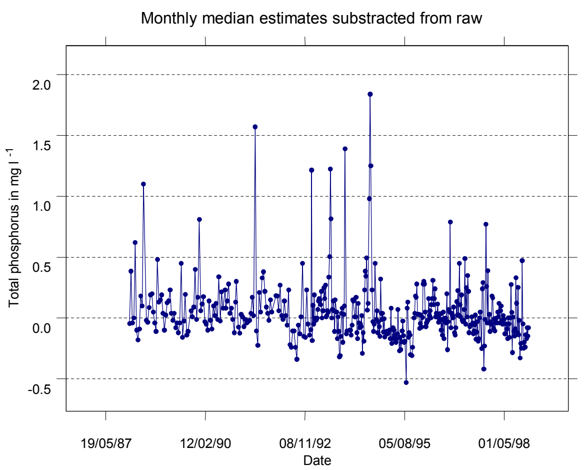
i) Raw data are pooled according to Julian day of sampling (years are ignored), in order to highlight the average shape of the intra-annual (i.e., between-month) variability. The median of the pooled values of each month is then estimated, and a first identification of outliers is performed at the same time (Figure A2.2.3.1).

Figure A2.2.3.1. Medians of the pooled values for each month of phosphorus at Eijsden for 1988–1998. These box plots provide a graphical illustration of the seasonal pattern, the main features of which are summer maximum values, and a local minimum in September. The circles in the boxes are monthly mean estimates; the horizontal bars inside the boxes are monthly median estimates. The circles outside the boxes are outlying values.



ii) The previous monthly median estimates are subtracted from the original data values, thus removing a robust estimate of the seasonal component of the time series (Figure A2.2.3.2).

Figure A2.2.3.2. Plot of the monthly median estimates subtracted from the raw data for phosphorus at Eijsden.

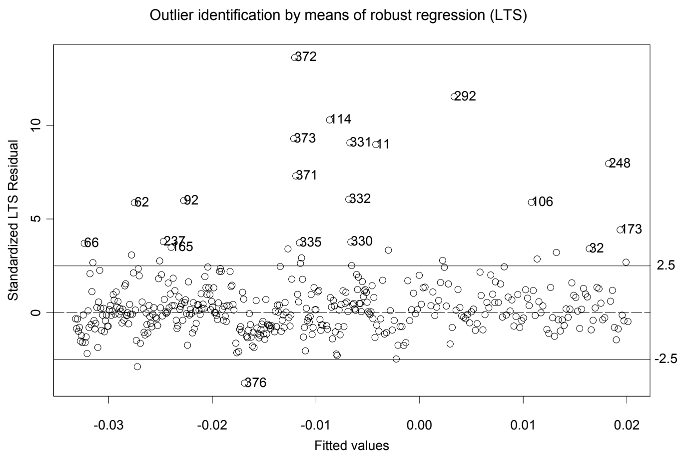


NB1: The existence of seasonality is known *a priori*, and no test is performed to assess its statistical significance.

NB2: In the following, the variability of monthly term estimates will be neglected; however, a comprehensive analysis should manage this uncertainty, especially by quantifying its effect on trend estimates (e.g., by resampling).

iii) A robust linear regression model (Least Trimmed Squares regression) is then fitted to the deseasonalized time series, in order to separate outliers from the bulk of observations (Figure A2.2.3.3), the latter exhibiting a linear trend "polluted" by serial correlation. The intention here is to identify outlying observations which obviously will not adequately fit to a simple model of trend (here a linear trend).

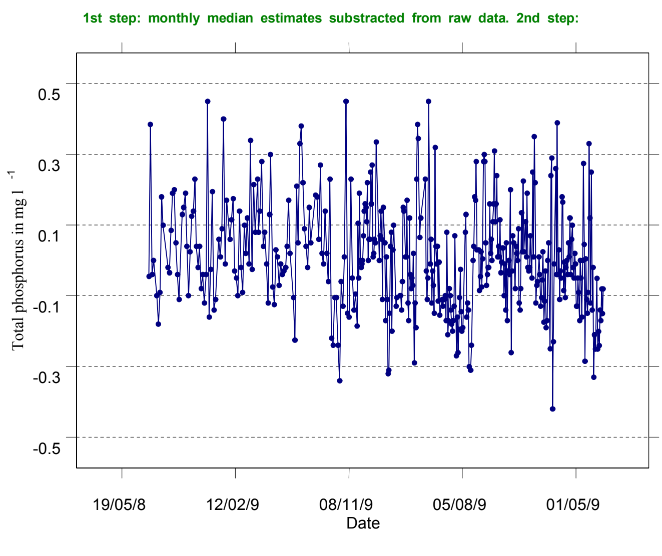
Figure A2.2.3.3. Plot of the outliers identified by fitting the Least Trimmed Squares regression model to the deseasonalized time series.



Twenty troublesome observations are thus removed; this amounts to approximately 10 % of the total number of measurements. From a phenomenological point of view, it is likely that these "extreme" values are highly informative. They will not be taken into account here, but it is worth emphasizing that explaining their occurrence should constitute a subject of study for biogeochemists.

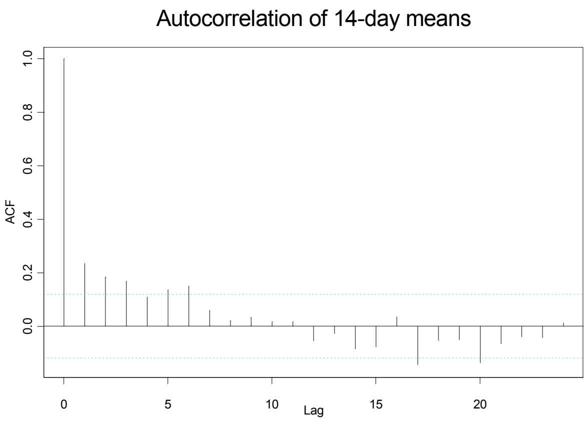
From now, attention will be paid to the "clean" deseasonalized time series only (Figure A2.2.3.4).

Figure A2.2.3.4. Deseasonalized time series of phosphorus at Eijsden, free from outlying values.



iv) Before estimating the autocorrelation function (ACF), a regularization of the sampling time step is performed, because the data have been collected at unequal time intervals. Some pooling over the shortest possible time scale is thus necessary, in order to build the average measurement values reproducing at best the data set that would have been obtained by regular sampling. A time step of two weeks has been retained after trial and error; the corresponding ACF estimate is shown in Figure A2.2.3.5.

Figure A2.2.3.5. ACF estimated for the time series shown in Figure A2.2.3.4, after aggregation and averaging of observations over consecutive time periods of fourteen days (avoiding the occurrence of "holes" in the series). Abscissa scale: lag in two-week units.



v) The ACF reveals a quite complex "memory" of the process, in accordance with the bumps observed on Figure A2.2.3.4. Nevertheless, for the sake of simplicity and, first of all, in order to manage a limited set of unknown parameters, an AR-1 model is postulated, i.e.,

$$\begin{aligned} u_t &= x_t - \rho x_{t-1} \ , \\ v_t &= y_t - \rho y_{t-1} \ , \\ \omega_t &= \varepsilon_t - \rho \varepsilon_{t-1} \end{aligned} \qquad [1]$$

where y, x, ε, and ρ, respectively, are the response variable, the regressor, the error term, and the first order autocorrelation coefficient (the index 1 designates the time increment). The linear model whose error term ω is free from first order serial correlation can be written as:

$$v_t = \beta_0 (1-\rho) + \beta_1 u_t + \omega_t \quad , \quad t = 1, \ldots, n \qquad [2]$$

with n = 259 in the present case. Estimation of the parameter β_1 gives the rate of change of y with x [rate expressed in 1/(2 weeks)], i.e., the trend.

2.4 Results and Discussion

Fitting model [1, 2] gives the following estimate b_1 of the slope β_1:

$b_1 = -4 \times 10^{-4}$, $SE(b_1) = 10^{-4}$, both expressed in mg l^{-1} P (14-day period)$^{-1}$,

computed t-statistic = –3.072,

with critical probability ≈ 2×10^{-3}.

It was further checked that the residuals in model [2] are no longer autocorrelated.

Roughly speaking, this result leads to a conclusion of a decrease of total phosphorus concentration in water at an average rate of the order of 10^{-2} mg l^{-1} P year^{-1}.

It should be noted that for a more detailed analysis several uncertainties remain to be taken into account, for instance, the sampling noise associated with the seasonality and also with the autocorrelation coefficient estimates; it is not easy to guess the amplitude of their impact on the stability of the previous estimator.

2.5 Statistical Analysis: Second Approach

2.5.1 Statistical Model

The second approach to analysing the phosphorus data at the station Eijsden is based on a linear regression method which accounts for the seasonality and autocorrelation in time. When time series data are used in regression analysis, often the error term is not independent through time. If the error term is autocorrelated, the efficiency of ordinary least-squares (OLS) parameter estimates is adversely affected and standard error estimates are biased. The Durbin-Watson d-statistic can be used to test for the absence of first-order autocorrelation in OLS residuals. If serial correlation is detected, estimation methods which account for the autocorrelation give better estimates.

A small drawback of this way of modelling is that only ordered and equally spaced time series data could be used with no missing values. Therefore, the data have been transformed to biweekly measurements. This model is defined in two parts: the structural term and the error term. For the latter term, an autoregressive process is assumed.

The model, which includes a linear trend and a seasonal component, is defined as:

$$y_t = \alpha + \beta_1 year + \beta_2 \sin\left(\frac{2\pi month}{12}\right) + \beta_3 \cos\left(\frac{2\pi month}{12}\right) + v_t \quad [3]$$

where y_t is the measured total phosphate at time t,

α is the intercept,

β_1 is the coefficient of the global linear year trend,

β_2 is the coefficient of the sinus component,

β_3 is the coefficient of the cosinus component,

year is the year value: 1988,…,1999,

month is the month value: 1,…12,

v_t is the error for time t.

The error is described as an autoregressive AR(p) process according to:

$$v_t = \varepsilon - \varphi_1 v_{t-1} - \ldots - \varphi_p v_{t-p}, \quad [4]$$

where ε_t is Normally and independently distributed with a mean of 0 and a variance of σ^2.

As this model incorporates a systematic part and the autocorrelation process of the time series, it could be useful for forecasting.

2.6 Analysis of the Data

In Figure A2.2.6.1, the results are presented for the biweekly time series. The residuals are analysed on the presence of autocorrelation. In the autocorrelation plot of the residuals of the least squares model (OLS), there appear two significant lags: 1 and 3. In the second figure, model 1 has been used which includes these autoregression coefficients and the autocorrelation has nearly vanished. It is nice to see that this model is capable of picking up some peaks (Figure A2.2.6.1). In Table A2.2.6.1, the coefficient β_1 and the standard error for the OLS model and the extended model are presented. It is clear that because of the autocorrelated errors, the estimate of the standard error for β_1 and hence the respective statistical test are biased.

Figure A2.2.6.1. The biweekly time series for the concentrations of total phosphorus (mg l^{-1}) at Eijsden. In the upper left corner, the values are fitted with an ordinary least squares (OLS) model which incorporates the linear trend and the seasonality. In the upper right corner, the residuals of this model are checked for autocorrelation. The dotted lines in this plot are the confidence limits of the autocorrelation with respect to H_0: = 0. In the lower left corner, the results are shown for the OLS model with the autocorrelated process included (AR(1,3)). In this figure the linear trend line is also included. The remaining residuals are shown in the lower right corner. No more autocorrelation can be detected.

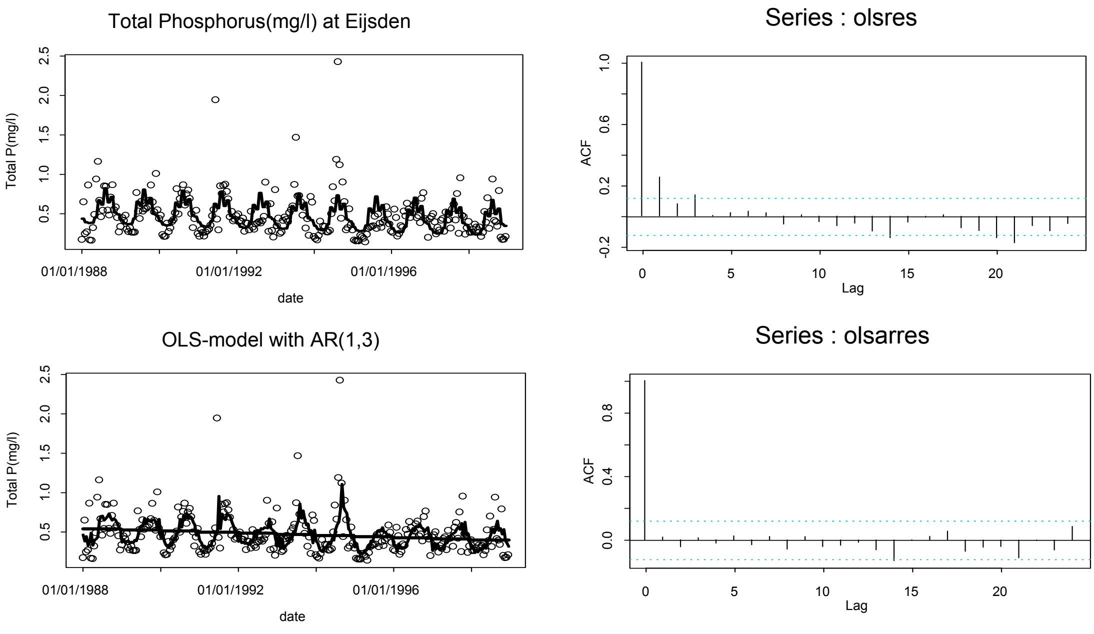


Table A2.2.6.1. The results of the regression analysis on the biweekly values, and the annual mean and median values, and the mean and median winter values. The OLS-model refers to the structural part of model 1; OLS+AR(1,3) refers to the full model 1 with the AR-part at lag 1 and 3 without the cosinus part; the OLS-Mean year is the linear trend model with the mean year values; the OLS-Mean winter year is the same model with the mean winter year values; the OLS-Median year is the OLS linear trend model with median year values; and the OLS-Median winter year is the same model with the median winter year values. Winter is defined as the three-month period from December to February.

Model	**N**	**beta_1 (mg l^{-1} $year^{-1}$)**	**Standard error**	**$P(T < \|t\|)$ $\|H_0$:beta_1 = 0)**
OLS-model	262	−0.0147	0.0044	0.010
OLS+AR(1,3)	262	−0.0144	0.0066	0.030
OLS-Mean year	11	−0.0132	0.0064	0.068
OLS-Median year	11	−0.0117	0.0049	0.049
OLS-Mean winter year	11	−0.0118	0.0051	0.044
OLS-Median winter year	11	−0.0091	0.0065	0.193

In Figure A2.2.6.2, the results are presented of the trend analysis of the annual mean and the annual median. This comparison should reflect the robustness of the median instead of the mean. It is clearly shown in Figure A2.2.6.1 that there are some peaks in the early years and also that the fluctuation is different during the years. Especially before 1995 the signal fluctuates much more. In Figure A2.2.6.2 this fluctuation becomes clearer where the annual mean, annual median, and the maximum and minimum are shown. The median is less fluctuating than the mean (see also Figure A2.2.6.3).

Figure A2.2.6.2. The annual mean, annual median, and the minimum and maximum values of total phosphorus (mg l^{-1}) at Eijsden.

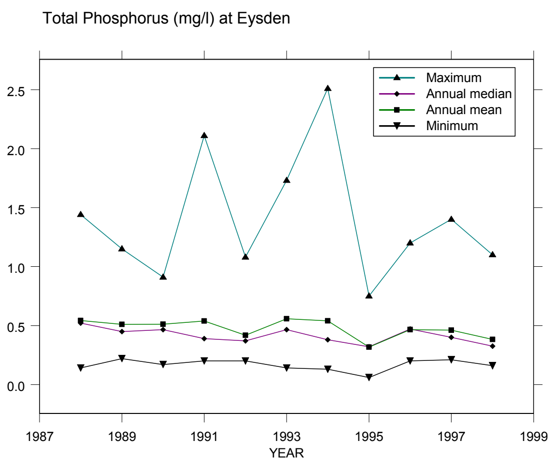
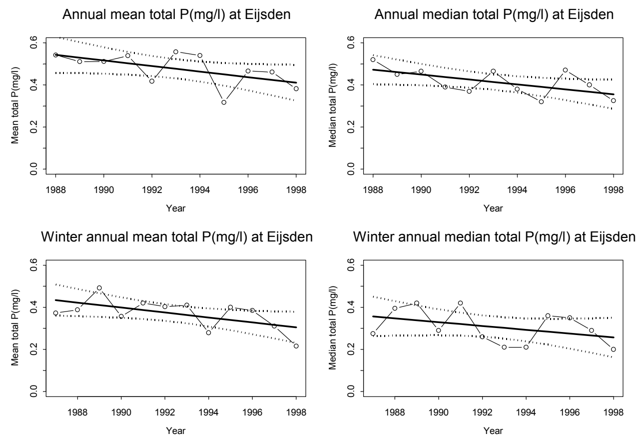


Figure A2.2.6.3. The linear trend lines of, respectively, the annual mean, annual median, winter annual mean, and winter annual median of total phosphorus (mg l^{-1}) with their respective 95 % confidence limits.



A winter period was also chosen, which is defined as the three-month period from December until February. The choice of the winter values is based on the fact that during this period the concentration of total phosphorus is not affected by biological activity, which caused part of the seasonal periodicity, and would be a more stable period for making a comparison each year. It is, however, not the period with the highest values (see Figure A2.2.2.1), which may rather be also a good period to choose for detecting a trend in the data. The results of the regression on the annual values and the winter values are presented in Table A2.2.6.1.

It is interesting to note that, except for the median winter values, the trend coefficients are nearly the same with the same standard errors. The bad behaviour of the median could be caused by the high autocorrelation of the winter values.

ANNEX 3

GEOHAB: SCOR-IOC PROGRAMME ON GLOBAL ECOLOGY AND OCEANOGRAPHY OF HARMFUL ALGAL BLOOMS

Proliferations of microalgae in marine or brackish waters can cause massive fish kills, contaminate seafood with toxins, and alter ecosystems in ways that humans perceive as harmful. The scientific community refers to these events with a generic term, "Harmful Algal Bloom" (HAB); while recognizing that some species cause toxic effects even at low cell densities, not all HAB species are technically "algae" or necessarily form dense "blooms". A broad classification of HAB species distinguishes two groups: the toxin producers, which can contaminate seafood or kill fish, and the high-biomass producers, which can cause anoxia and indiscriminate kills of marine life after reaching dense concentrations. Some HAB species have characteristics of both.

Although HABs occurred long before human activities began to transform coastal ecosystems, a survey of affected regions and of economic losses and human poisonings throughout the world demonstrates very well that there has been a dramatic increase in the impacts of HABs over the past few decades. The HAB problem is now widespread and serious. It must be remembered that the harmful effects of HABs extend well beyond direct economic losses and impacts on human health. When HABs contaminate or destroy coastal resources, the livelihoods of local residents are threatened and the sustenance of human populations is compromised. Clearly, there is a pressing need to develop effective responses to the threat of HABs through management and mitigation. This requires knowledge of the factors that control the distributions and population dynamics of HAB species.

A great deal is known about harmful algae and HABs, but the existing ability to describe the factors controlling the dynamics of individual species is limited by critical gaps in knowledge about how the physiological, behavioural, and morphological characteristics of algae (including HAB species) interact with environmental conditions to promote the selection for one species vs. another. For example, information about the environmental cues for encystment (formation of a resting stage) and germination, as well as the interactions of life cycles with hydrography, is generally inadequate to quantify the role of resting stages in the population dynamics of cyst-forming harmful algae. Also, it is often difficult to assess the role of nutrients and light in algal population dynamics and toxicity because some phytoplankton can migrate vertically to exploit deep sources of nutrients at night and light near the surface during the day. Further, some harmful species can exploit several forms of nutrition (including consumption of other microorganisms), thereby complicating models of growth or toxicity vs. nutrient concentration. Compounding the problems, essentially all effects of physical forcing and nutrient supply on harmful algal populations also influence food-web/community interactions, which ultimately determine the selection for or against a particular species.

Successful research to date shows that the key to explaining HAB phenomena is to identify and quantify special adaptations of HAB species that lead to their selection in particular hydrodynamic and ecological conditions. Thus, the central research problem for GEOHAB is to understand the critical features and mechanisms underlying the population dynamics of HAB species in a variety of oceanographic regimes. This understanding can be used as a basis for monitoring and predicting the occurrence, movement, toxicity, and environmental effects of harmful algal blooms. In turn, monitoring and prediction are essential for management and mitigation of HABs.

Since HAB species are found in marine and brackish-water ecosystems worldwide, the central research problem can be addressed comprehensively and effectively only through international, interdisciplinary, and comparative research on the dynamics of HABs within their oceanographic and ecological systems. Progress depends upon advancement through targeted studies and technological innovation in biology, ecology, chemical and physical oceanography, modelling, and ocean observation.

The SCOR-IOC Programme on Global Ecology and Oceanography of Harmful Algal Blooms (GEOHAB) was established to address the need for broad-based advancement in the understanding of HABs. The mission of GEOHAB is to:

"*foster international cooperative research on HABs in ecosystem types sharing common features, comparing the key species involved and the oceanographic processes that influence their population dynamics*."

The scientific goal of GEOHAB is to:

"improve prediction of HABs by determining the ecological and oceanographic mechanisms underlying the population dynamics of harmful algae, integrating biological, chemical and physical studies supported by enhanced observation and modelling systems."

The scientific goal of GEOHAB will be approached by addressing research questions such as:

1) What are the environmental factors that determine the changing distribution of HAB species, their genetic variability, and the biodiversity of associated communities?

2) What are the unique adaptations of HAB species that determine when and where they occur and the extent to which they produce harmful effects?

3) What are the effects of increasing human activities (e.g., eutrophication, translocation of species) on the occurrence of HABs?

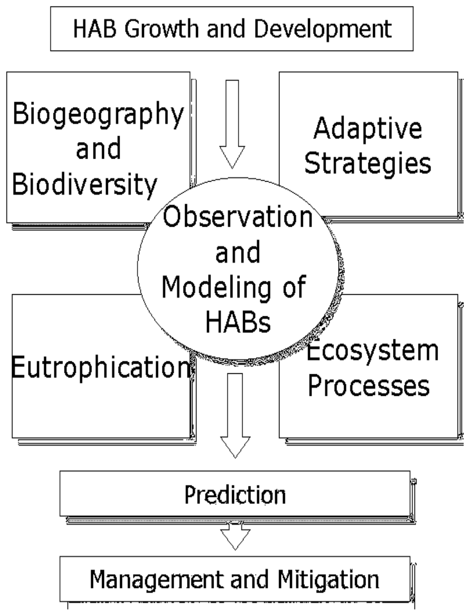
4) How do HAB species, their population dynamics and community interactions respond to changes in their environment?

A broad range of research is directly relevant to these questions, including: interdisciplinary process studies of HABs in comparable ecosystems; taxonomic and genetic surveys of HAB organisms from different locations, along with physiological characterization of isolates and examination of possible dispersal mechanisms; studies of the influences of turbulence or variations in nutrient fluxes on community interactions (conducted in microcosms or mesocosms, modelled, and compared with observations of natural communities); and examination of temporal and spatial trends in phytoplankton dynamics (including HABs) relative to human influence and climate variability, as inferred from existing data sets and the developing Global Ocean Observing System (GOOS).

Targeted studies and technological innovation are essential to the GEOHAB programme, and there are many opportunities for advancement. For example, better tools for detecting harmful algae and their biologically active products and more sensitive approaches for studying the nutrition of HAB organisms are needed. Also required are integrated techniques for observing physical, chemical, and biological variability on the scales relevant to physical forcing and more effective observation systems for detecting and characterizing microalgal population dynamics (including HAB species). Improved representations of the physical processes that influence HABs are dependent upon progress in physical-biological coupled models (including data assimilation). Significant advances in biotechnology and instrumentation for measuring physical and bio-optical variability in the sea together with improvements in computational abilities, ensure that rapid progress can be made.

The GEOHAB Scientific Steering Committee is charged with identifying the scientific issues and detailed goals and objectives for an international study of the ecology and oceanography of harmful algal blooms which builds upon a broad range of national and regional efforts. It is not the intention to specify or circumscribe the research directions for national HAB research activities. GEOHAB will foster scientific advancement in the understanding of HABs by encouraging and coordinating fundamental scientific research—multifaceted, international, and interdisciplinary—maintaining an ecological and oceanographic context consistent with the scientific goal of the programme. International, cooperative research on comparable ecosystems would be encouraged. In addition, GEOHAB will identify targeted studies on organisms, processes, methods, and observation technologies that are needed to support the interdisciplinary research. Improved global observation systems are required to resolve influences of environmental factors (anthropogenic and climate-related) on distributions and trends in HAB occurrence. This will be greatly facilitated through strong links between GEOHAB and the Global Ocean Observing System (GOOS). The organization of the GEOHAB programme elements is shown in Figure A3.1.

Figure A3.1. Organization of the GEOHAB Programme elements.



A better understanding of the factors that regulate the dynamics of HABs in the context of physical and chemical forcing, ecosystem dynamics, and human influences will be used to improve strategies for monitoring and prediction of HABs. The benefits of GEOHAB will be better methodologies for predicting the occurrence, distribution, toxicity, and other deleterious environmental effects of HABs. However, this is not the only benefit of GEOHAB. Through links to national agencies and international organizations responsible for protecting coastal resources and public health, the knowledge gained from GEOHAB will be used to develop international capabilities for more effective management and mitigation of HAB problems. This linking of basic scientific research directly to societal needs should result in an effective contribution of science to the protection of the intrinsic and economic value of coastal marine ecosystems and their management.

ANNEX 4

CERTIFIED REFERENCE MATERIALS FOR ORGANIC COMPOUNDS FOR USE IN MARINE MONITORING

This annex contains a series of tables describing current certified reference materials (CRMs) for organic contaminants that are of use for marine monitoring programmes. The following comments apply to the tables in this annex:

- the compiled tables are for information. Although every effort has been made to ensure that these tables are accurate, users of CRMs should consult vendors for full and accurate information;
- certified calibration materials and standards are not included;
- these tables do not purport to be complete and all the CRMs listed may not be commercially available;
- methylmercury is not considered an organic contaminant for the purposes of this list.

The suppliers and their websites are as follows:

- NIST: USA, National Institute of Standards and Technology [http://ois.nist.gov/srmcatalog];
- IAEA: UN, International Atomic Energy Agency; [http://www.iaea.org/programmes/naal/pci/pages/aqcs.htm];
- BCR: EC, Bureau of Community Reference, now EC Institute for Reference Materials and Measurements (IRMM) [http://www.irmm.jrc.be/rm];
- NRC: Canada, National Research Council, Institute for National Measurement Standards (INMS) [http://www.imb.nrc.ca/crmp_e.html];
- NWRI: Canada, National Water Research Institute, Environment Canada [http://www.cciw.ca/nwri/nwri.html];
- CIL: USA, Cambridge Isotope Laboratories [http://www.isotope.com/newcat.htm];
- NIES: Japan, National Institute for Environmental Standards, Japan Environment Agency [http://www.nies.go.jp].

Table A4.1a. Reference materials for organic contaminants in marine biota. Values preceded by an asterisk (*) are non-certified; all other values are certified.

Code	**SRM-1974a**	**SRM-2974**	**140/OC**
Organization	**SRM-NIST**	**SRM-NIST**	**IAEA**
Country of origin	USA	USA	Monaco
Matrix	Mussel tissue	Mussel tissue	*Fucus* (sea plant homogenate)
UNITS	µg kg^{-1}	µg kg^{-1}	µg kg^{-1}
AS	Dry weight	Dry weight	Dry weight
[±] ***expressed as***	± 95% CI	± 95% CI	± 95% CI
UNITS OF ISSUE	3 × 15 g	8 g	35 g
FORM	Frozen	Freeze-dried	Freeze-dried
Hydrocarbons	µg kg^{-1}	µg kg^{-1}	µg kg^{-1}
Resolved aliphatics			*13 mg kg^{-1}
Unresolved aliphatics			*26 mg kg^{-1}
Pristane			50
Phytane			56
Sum alkanes (C14–C34)			11 mg kg^{-1}
Total aromatics			*5.8 mg kg^{-1}
Unresolved aromatics			*0.35 mg kg^{-1}
Anthracene	6.1 ± 1.7	6.1 ± 1.7	14
Benz[*a*]anthracene	32.5 ± 4.7	32.5 ± 4.8	25
Benzo[*b*]fluoranthene	46.4 ± 3.7	46.4 ± 4.0	*37
Benzo[*k*]fluoranthene	20.18 ± 0.84	20.2 ± 1.0	19
Benzo[*a*]pyrene	15.63 ± 0.65	15.63 ± 0.80	20
Benzo[*e*]pyrene	84.0 ± 1.9	84.0 ± 3.2	26
Benzo[*ghi*]perylene	22.0 ± 2.2	22.0 ± 2.3	20
Chrysene	44.2 ± 2.3	44.2 ± 2.7	40
Dibenz[*a,h*]anthracene			*4.5
Fluoranthene	163.7 ± 9.1	163.7 ± 10.3	88
Fluorene			*6.5
Indeno[1,2,3-*c,d*]pyrene	14.2 ± 2.8	14.2 ± 2.8	33
1-Methylphenanthrene			11
2-Methylphenanthrene			19
Naphthalene	23.5 ± 4.4		17
Perylene	7.68 ± 0.27	7.68 ± 0.35	
Phenanthrene	22.2 ± 2.4	22.2 ± 2.5	76
Pyrene	151.6 ± 6.6	151.6 ± 8.0	67
Triphenylene	50.7 ± 5.9	50.7 ± 6.1	
Total aliphatics			*27 mg kg^{-1}
***n*-C17**			890
***n*-C18**			99
UVF Chrysene			*3.5 mg kg^{-1}
UVF ROPME oil			*29 mg kg^{-1}

Table A4.1b. Reference materials for organic contaminants in marine biota. Values preceded by an asterisk (*) are non-certified; all other values are certified.

Code	**SRM-1974a**	**SRM-1588a**	**SRM-1945**	**SRM-2974**	**MA-A-3/OC**	**MA-A-1/OC**	**140/OC**	**BCR 598**
Organization	**SRM-NIST**	**SRM-NIST**	**SRM-NIST**	**SRM-NIST**	**IAEA**	**IAEA**	**IAEA**	**BCR**
Country of origin	USA	USA	USA	USA	Monaco	Monaco	Monaco	EC
Matrix	Mussel tissue	Cod liver oil	Whale blubber	Mussel tissue	Shrimp homogenate	Copepoda	*Fucus* (sea plant homogenate)	Cod liver oil
UNITS	µg kg^{-1}	µg kg^{-1}	µg kg^{-1}	µg kg^{-1}	ng g^{-1}	ng g^{-1}	µg kg^{-1}	µg kg^{-1}
AS	Dry weight		Wet weight	Dry weight	Dry weight	Dry weight	Dry weight	Wet weight
[±] ***expressed as***	± 95% CI	± 95% CI	± 95% CI	± 95% CI	± 95% CI	± 95% CI	95% CI	95% CI
UNITS OF ISSUE	3 × 15 g	5 × 1.2 ml/ ampoule	Set 2, 15 g/ ampoule	8 g	35 g	35 g	30 g	5 g
FORM	Frozen	Oil	Frozen	Freeze-dried	Freeze-dried	Freeze-dried	Freeze-dried	Freeze-dried
PESTICIDES								
Hexachlorobenzene		157.8 ± 5.0	32.9 ± 1.7		0.32 (0.2–0.44)		*1.3	55.7 ± 2.0
α-HCH		85.3 ± 3.4	16.2 ± 3.4		15	10 (1.6–18.4)	*1.4	42 ± 3
β-HCH			*8.0 ± 1.4				4.6	16 ± 3
γ-HCH		24.9 ± 1.7	3.30 ± 0.81		3.2	8.2	*11	23 ± 4
Aldrin					0.7 (0.2–1.2)	14 (0–33)	*0.76	
***trans*-Chlordane**	16.6 ± 1.7			16.6 ± 1.8				6.9 ± 1.6
***cis*-Chlordane**	17.2 ± 2.8	167.0 ± 5.0	46.9 ± 2.8	17.2 ± 2.9			*1.4	24.4 ± 1.8
Heptachlor					2.4		*3	
Heptachlor epoxide		31.6 ± 1.5	10.8 ± 1.3				*0.79	
***trans*-Nonachlor**	18.0 ± 3.6	214.6 ± 7.9	231 ± 11	18 ± 3.6				39 ± 4
Dieldrin	*6.2 ± 1.3	155.9 ± 4.5	*37.5 ± 3.9				1.7	59 ± 4
***cis*-Nonachlor**	6.84 ± 0.90	94.8 ± 2.8	48.7 ± 7.6	6.84 ± 0.92				
Oxychlordane			19.8 ± 1.9					11.0 ± 1.8
2,4′-DDE	*5.26 ± 0.27	22.0 ± 1.0	12.28 ± 0.87					
4,4′-DDE	51.2 ± 5.5	651 ± 11	445 ± 37	51.2 ± 5.7	4.7 (1.3–8.1)	6.1 (1.5–17.1)	1.2	610 ± 40
2,4′-DDD	*13.7 ± 2.8	36.3 ± 1.4	18.1 ± 2.8					30 ± 4
4,4′-DDD	43.0 ± 6.3	254 ± 11	133 ± 10	43 ± 6.4	0.81 (0.05–1.57)		0.7	400 ± 30
2,4′-DDT	*8.5 ± 1.9	156.0 ± 4.4	106 ± 14					
4,4′-DDT	3.91 ± 0.59	524 ± 12	245 ± 15	3.91 ± 0.60	3.2 (0–6.7)	8.3 (3.4–13.2)	2.2	179 ± 18
Mirex			28.9 ± 2.8					
Endrin							*0.71	
α-Endosulfan							*0.9	
Total PCBs								
Aroclor 1242						120 (67–173)		
Aroclor 1254					33 (0–67)	140 (70–210)	*25	
Aroclor 1260							*12	

Table A4.1c. Reference materials for organic contaminants in marine biota. Values preceded by an asterisk (*) are non-certified; all other values are certified.

Code	**SRM-1974a**	**SRM-1588a**	**SRM-1945**	**SRM-2974**	**140/OC**	**CARP-1**	**BCR 349**	**BCR 350**	**EDF-2525**	**EDF-2526**
Organization	**SRM-NIST**	**SRM-NIST**	**SRM-NIST**	**SRM-NIST**	**IAEA**	**NRC**	**BCR**	**BCR**	**CIL**	**CIL**
Country of origin	USA	USA	USA	USA	Monaco	Canada	EC	EC	USA	USA
Matrix	Mussel tissue	Cod liver oil	Whale blubber	Mussel tissue	*Fucus* (sea plant homogenate)	Common carp	Cod liver oil	Mackerel oil	Fish	Fortified fish
UNITS	μg kg^{-1}	μg kg^{-1}	μg kg^{-1}	μg kg^{-1}	μg kg^{-1}	ng kg^{-1}	μg kg^{-1}	μg kg^{-1}	ng kg^{-1}	ng kg^{-1}
AS	Dry weight	Wet weight	Wet weight	Dry weight	Dry weight	Wet weight	Wet weight	Wet weight	Wet weight	Wet weight
[±] ***expressed as***	± 95% CI	± 95% CI	± 95% CI	± 95% CI	95% CI	± 95% CI	± 95% CI	± 95% CI	95% CI	95% CI
UNITS OF ISSUE	3 × 15 g	5 × 1.2 ml/ ampoule	Set 2, 15 g/ ampoule	8 g	35 g	6 × 9 g	2 g ampoules	2 g ampoules	Set 1, 10 g/ ampoule	Set 1, 10 g/ ampoule
FORM	Frozen	Oil	Frozen	Freeze-dried	Freeze-dried	Slurry	Oil	Oil	Slurry	Slurry
PCBs						μg kg^{-1}				
PCB 18	*33 ± 11		4.48 ± 0.88							
PCB 28	*79 ± 15	28.32 ± 0.55	*14.1 ± 1.4		1.7		68 ± 7	22.5 ± 4.0		
PCB 31	*76 ± 21	8.33 ± 0.28	*3.12 ± 0.69							
PCB 32					1.8					
PCB 44	72.2 ± 7.4	35.1 ± 1.4	12.2 ± 1.4	72.7 ± 7.7			*75	*44		
PCB 49	88.8 ± 5.0	29.9 ± 0.84	20.8 ± 2.8	88.8 ± 5.7	1.6					
PCB 52	115 ± 11	83.3 ± 2.3	43.6 ± 2.5	115 ± 12	3.8	124 ± 32	149 ± 20	62 ± 9		
PCB 66	101.4 ± 4.4	54.7 ± 1.5	23.6 ± 1.6	101.4 ± 5.4						
PCB 77									1945 ± 354	619 ± 107
PCB 87	54 14*	56.3 ± 1.1	16.7 ± 1.4							
PCB 95	83 ± 17	36.5 ± 1.1	33.8 ± 1.7	83 ± 17						
PCB 99	70.9 ± 4.0		45.4 ± 5.4	70.9 ± 4.5						
PCB 101	128.3 ± 9.7	126.5 ± 4.3		128 ± 10	2.4		370 ±17	165 ± 9		
PCB 101/90			65.2 ± 5.6			124 ± 37				
PCB 105	53.0 ± 3.4	60.2 ± 2.3	30.1 ± 2.3	53 ± 3.8	0.49	54 ± 24				
PCB 110	127.3 ± 8.6	76.0 ± 2.0	23.3 ± 4.0	127.3 ± 9.4						
PCB 118	130.8 ± 3.6	176.3 ± 3.8	74.6 ± 5.1	130.8 ± 5.3	1	132 ± 60	456 ± 31	143 ± 20		
PCB 126									647 ± 148	1140 ± 465
PCB 128	22.0 ± 3.4	47.0 ± 2.4	23.7 ± 1.7	22 ± 3.5			*104	*41		
PCB 138					1.7					
PCB 138/163							*765	*274		
PCB 138/163/164	133.5 ± 9.5	263.5 ± 9.1	131.5 ± 7.4	134 ± 10		102 ± 23				
PCB 149	87.6 ± 2.3	105.7 ± 3.6	106.6 ± 8.4	87.6 ± 3.5	1.2					
PCB 151	25.6 ± 3.5	54.8 ± 2.1	28.7 ± 5.2	25.6 ± 3.6						
PCB 153	145.2 ± 7.6	273.8 ± 7.7	213 ± 13	145.2 ± 8.8	1.7	83 ± 39	938 ± 40	317 ± 20		
PCB 156	7.43 ± 0.99	27.3 ± 1.8	10.3 ± 1.1	7.4 ± 1.0	0.17					
PCB 169									50 ± 12	1416 ± 553
PCB 170	5.5 ± 1.1	46.5 ± 1.1		5.5 ± 1.1	*0.21					
PCB 170/190			40.6 ± 2.6			22 ± 8				
PCB 180	17.1 ± 3.8	105.0 ± 5.2	106.7 ± 5.3	17.1 ± 3.8	0.43	46 ± 14	280 ± 22	73 ± 13		
PCB 183	16.0 ± 2.4	31.21 ± 0.62	36.6 ± 4.1	16.0 ± 2.4						
PCB 187			105.1 ± 9.1							
PCB 187/182						36 ± 16				
PCB 187/159/182	34.0 ± 2.3	35.23 ± 0.83		34.0 ± 2.5						
PCB 194		15.37 ± 0.61	39.6 ± 2.5				*38			
PCB 195			17.7 ± 4.3							
PCB 201		12.18 ± 0.46	16.96 ± 0.89							
PCB 206			31.1 ± 2.7							
PCB 209			10.6 ± 1.1							

Table A4.1d. Reference materials for organic contaminants in marine biota. Values preceded by an asterisk (*) are non-certified; all other values are certified.

Code	**SRM-1588a**	**CARP-1**	**BCR 477**	**NIES11**	**EDF-2525**	**EDF-2526**
Organization	**SRM-NIST**	**NRC**	**BCR**	**NIES**	**CIL**	**CIL**
Country of origin	USA	Canada	EC	Japan	USA	USA
Matrix	Cod liver oil	Common carp	Mussel tissue	Fish tissue	Fish	Fortified fish
UNITS	$\mu g\ kg^{-1}$	$ng\ kg^{-1}$	$mg\ kg^{-1}$	$\mu g\ g^{-1}$	$ng\ kg^{-1}$	$ng\ kg^{-1}$
AS	Wet weight	Wet weight	Dry weight	Dry weight	Wet weight	Wet weight
[±] ***expressed as***	± 95% CI	± 95% CI	½-width of 95% CI of mean	2 × SD	95% CI	95% CI
UNITS OF ISSUE	5 × 1.2 ml/ ampoule	6 × 9 g	14 g	20 g	Set 1, 10 g/ ampoule	Set 1, 10 g/ ampoule
FORM	Oil	Slurry	Freeze-dried	Freeze-dried	Slurry	Slurry
Dioxins and furans						
2,3,7,8 - TCDF		11.9 ± 2.7			22 ± 1.6	17 ± 1.5
1,2,3,7,8 - PCDF		5.0 ± 2.0			4.9 ± 0.56	40 ± 3.7
2,3,4,7,8 - PCDF					14 ± 1.3	38 ± 3.5
1,2,3,4,7,8 - HxCDF					8.2 ± 3.7	80 ± 8.4
1,2,3,6,7,8 - HxCDF					2.7 ± 1.2	63 ± 5.5
1,2,3,7,8,9 - HxCDF					0.76 ± 0.35	58 ± 7.0
2,3,4,6,7,8 - HxCDF					2.3 ± 1.9	60 ± 5.5
1,2,3,4,6,7,8 - HpCDF					4.4 ± 6.0	83 ± 9.2
1,2,3,4,7,8,9 - HpCDF					0.63 ± 0.23	73 ± 7.7
OCDF	*1.00				2.6 ± 1.3	190 ± 22
1,2,7 - TCDD	*0.32					
1,2,3,4 - TeCDD	*0.38					
2,3,7,8 - TCDD	*0.21	6.6 ± 0.6			17 ± 1.4	19 ± 1.4
1,2,3,7,8 - PCDD		4.4 ± 1.1			4.0 ± 0.57	40 ± 3.0
1,2,3,4,7,8 - HxCDD		1.9 ± 0.7			0.77 ± 0.27	60 ± 4.8
1,2,3,6,7,8 - HxCDD	*0.39	5.6 ± 1.3			3.0 ± 1.2	56 ± 4.8
1,2,3,7,8,9 - HxCDD	*0.22	0.7 ± 0.4			0.79 ± 0.26	60 ± 4.4
1,2,3,4,6,7,8 - HpCDD		6.5 ± 1.8			1.4 ± 0.53	76 ± 5.9
OCDD	*1.01	6.3 ± 1.9			7.2 ± 3.7	192 ± 14
Alpha-tocopherol	*134.2 ± 2.5					
Antifoulants						
Triphenyltin (as chloride)				*6.3		
Tributyltin (TBT)			2.2 ± 0.19	1.3 ± 0.1		
Dibutyltin (DBT)			1.54 ± 0.12			
Monobutyltin (MBT)			1.50 ± 0.28			

Table A4.2a. Reference materials for organic contaminants in marine sediments. Values preceded by an asterisk (*) are non-certified; all other values are certified.

Code	SES-1	HS-3B	HS-4B	HS-5	HS-6	SRM 1944	EC-1	EC-4	EC-8	BCR-535
Organization	**NRC-CNRC**	**NRC-CNRC**	**NRC-CNRC**	**NRC-CNRC**	**NRC-CNRC**	**SRM-NIST**	**NWRI**	**NWRI**	**NWRI**	**BCR**
Country of origin	Canada	Canada	Canada	Canada	Canada	USA	Canada	Canada	Canada	EU
Matrix	Estuarine sediment	Harbour sediment	Harbour sediment	Harbour sediment	Harbour sediment	New York New Jersey Waterway sediment	Freshwater harbour sediment	Freshwater harbour sediment	Lake sediment	Freshwater harbour sediment
UNITS	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	µg g^{-1}	µg g^{-1}	µg g^{-1}	mg kg^{-1}
AS	Dry weight	Dry weight	Dry weight	Dry weight	Dry weight	Dry weight	Dry weight	Dry weight	Dry weight	Dry weight
[±] ***expressed as***		90% CI	90% CI	90% CI	90% CI	95% Cl	±SD	±SD	±SD	½-width of 95% CI of mean
UNITS OF ISSUE	200 g	100 g	100 g	200 g	200 g	50 g	100 g	100 g	100 g	40 g
FORM	Freeze-dried	Freeze-dried	Freeze-dried	Freeze-dried	Freeze-dried	Freeze-dried	Freeze-dried	Freeze-dried	Freeze-dried	Dried
PAHs										
Acenaphthene	*7.21	1.25 ± 0.02	0.09 ± 0.02	0.23 ± 0.10	0.23 ± 0.07	*0.57± 0.03		*0.032	*0.013	
Acenaphthylene		0.6 ± 0.10	0.3 ± 0.10	<0.15	0.19 ± 0.05			*0.048	*0.028	
Anthracene	*1.63	2.76 ± 0.06	0.46 ± 0.06	0.38 ± 0.15	1.1 ± 0.4	1.77 ± 0.33	1.2	*0.124	*0.041	
Benzo[*a*]anthracene	*1.31	7.91 ± 0.09	1.46 ± 0.09	2.9 ± 1.2	1.8 ± 0.3	4.72 ± 0.11	8.7	*0.712	*0.168	1.54 ± 0.10
Benzo[*b*]chrysene						0.63 ± 0.10				
Benzo[*a*]fluoranthene						0.78 ± 0.12				
Benzo[*b*]fluoranthene				2.0 ± 0.15	2.8 ± 0.6	3.87 ± 0.42	7.8	*0.753	*0.208	2.29 ± 0.15
Benzo[*k*]fluoranthene				1.0 ± 0.4	1.43 ± 0.15	2.30 ± 0.20	4.4	*0.560	*0.294	1.09 ± 0.15
Benzo[*b,k*]fluoranthene (combined value)		12.8 ± 0.12	3.32 ± 0.12							
Benzo[*j*]fluoranthene						2.09 ± 0.44				
Benzo[*a*]pyrene	*1.21	5.80 ± 0.15	1.55 ± 0.15	1.7 ± 0.8	2.2 ± 0.4	4.30 ± 0.13	5.3	*0.675	*0.207	1.16 ± 0.10
Benzo[*e*]pyrene						3.28 ± 0.11	5.3	*0.747	*0.531	1.86 ± 0.13
Benzo[*g,h,i*]perylene	*1.21	3.88 ± 0.15	1.23 ± 0.15	1.3 ± 0.3	1.78 ± 0.72	2.84 ± 0.10	4.9	*0.576	*0.176	
Biphenyl		0.41 ± 0.02	0.04 ± 0.002			*0.32 ± 0.07				
Chrysene/Triphenylene							*9.2	*1.073	*0.378	
Chrysene	*1.32	8.77 ± 0.11	1.76 ± 0.11	2.8 ± 0.9	2.0 ± 0.3	4.86 ± 0.10				
Coronene		0.83 ± 0.04	0.31 ± 0.04							
Dibenz[*a,h*]anthracene	*1.30	0.89 ± 0.04	0.34 ± 0.04	0.2 ± 0.1	0.49 ± 0.16	0.424 ± 0.069	*1.3	*0.241	*0.316	
Dibenz[*a,c*]anthracene						0.335 ± 0.013				
Dibenz[*a,j*]anthracene						0.50 ± 0.044				
Dibenz*[a,i/*pyrene		0.59 ± 0.03	0.16 ± 0.03							
Dibenzofuran		2.2 ± 0.02	0.14 ± 0.02							
Dibenzothiophene		1.19 ± 0.02	0.11 ± 0.02			*0.62 ± 0.01				
Fluoranthene	*1.58	25.33 ± 0.11	3.33 ± 0.11	8.4 ± 2.6	3.54 ± 0.65	8.92 ± 0.32	23.2	*1.087	*0.462	
Fluorene	*1.42	2.38 ± 0.04	0.16 ± 0.04	0.4 ± 0.1	0.47 ± 0.12	*0.85 ± 0.03		*0.088	*0.019	
Indeno[1,2,3-*c,d*]pyrene	*1.28			1.3 ± 0.7	1.95 ± 0.58	2.78 ± 0.10	5.7	*0.564	*0.034	1.56 ± 0.14
1-Methylphenanthrene						*1.7 ± 0.1				
Naphthalene	*3.62	2.14 ± 0.02	0.22 ± 0.02	0.25 ± 0.07	4.1 ± 1.1	1.65 ± 0.31		*0.058	*0.01	
Perylene						1.17 ± 0.24	*1.1	*0.28	*0.202	
Phenanthrene	*1.37	18.8 ± 0.08	1.91 ± 0.08	5.2 ± 1.0	3.0 ± 0.6	5.27 ± 0.22	15.8	*0.732	*0.234	
Pyrene	*4.09	18 ± 0.10	2.55 ± 0.10	5.8 ± 1.8	3.0 ± 0.6	9.70 ± 0.42	16.7	*1.085	*0.327	2.52 ± 0.18
Picene						0.518 ± 0.093				
Triphenylene						1.04 ± 0.27				
Benzo[*c*]phenathrene						0.76 ± 0.10				
Pentaphene						0.288 ± 0.026				
1-Methylnaphthalene		0.73 ± 0.02	0.16 ± 0.02			*0.52 ± 0.08				

Table A4.2b. Reference materials for organic contaminants in marine sediments. Values preceded by an asterisk (*) are non-certified; all other values are certified.

Code	CS-1	HS-1	HS-2	SRM 1944	EC-1	EC-4	EC-8	BCR-536
Organization	NRC-CNRC	NRC-CNRC	NRC-CNRC	SRM-NIST	NWRI	NWRI	NWRI	BCR
Country of origin	Canada	Canada	Canada	USA	Canada	Canada	Canada	EU
Matrix	Harbour sediment	Harbour sediment	Harbour sediment	New York New Jersey Waterway sediment	Freshwater harbour sediment	Freshwater harbour sediment	Lake sediment	Freshwater harbour sediment
UNITS	$\mu g\ kg^{-1}$	$\mu g\ kg^{-1}$	$\mu g\ kg^{-1}$	$mg\ kg^{-1}$	$\mu g\ g^{-1}$	$\mu g\ g^{-1}$	$\mu g\ g^{-1}$	$\mu g\ kg^{-1}$
AS	Dry weight	Dry weight	Dry weight	Dry weight	Dry weight	Dry weight	Dry weight	Dry weight
[±] *expressed as*	± SD	± SD	± SD	95% CI	± SD	± SD	± SD	½-width of 95% CI of mean
UNITS OF ISSUE	200 g	200 g	200 g	50 g	100 g	100 g	100 g	40 g
FORM	Freeze-dried	Freeze-dried	Freeze-dried	Freeze-dried	Freeze-dried	Freeze-dried	Freeze-dried	Dried
Pesticides				$\mu g\ kg^{-1}$	$ng\ g^{-1}$	$ng\ g^{-1}$	$ng\ g^{-1}$	
Hexachlorobenzene				6.03 ± 0.35	*5.4	*2.2	*97.50	
***cis*-Chlordane**				16.51 ± 0.83				
***trans*-Nonachlor**				8.20 ± 0.51				
4,4′-DDT				119 ± 11				
PCBs				$\mu g\ kg^{-1}$	$ng\ g^{-1}$	$ng\ g^{-1}$	$ng\ g^{-1}$	
PCB 8				22.3 ± 2.3				
PCB 18				51.0 ± 2.6	*47.4 ± 16.1	*3.7 ± 1.6	*2.1	
PCB 28				80.0 ± 2.7	*48.7 ± 17.0	*6.8 ± 1.8	*5.2	44 ± 5
PCB 31				78.7 ± 1.6[1]				
PCB 44				60.2 ± 2.0	*64.7 ± 31.4	*7.5 ± 2.9	*10.4	
PCB 49				53.0 ± 1.7				
PCB 52				79.4 ± 2.0	*99.4 ± 43.2	*12.5 ± 5.7	*14.2	38 ± 4
PCB 66				71.9 ± 4.3				
PCB 77					*4.1			
PCB 87				29.9 ± 4.3	*44.9 ± 14.5	*8.3 ± 1.5	*6.9	
PCB 95				65.0 ± 8.9				
PCB 99				37.5 ± 2.4				
PCB 101/90				73.4 ± 2.5				
PCB 101		1.62 ± 0.21	5.42 ± 0.34		*109.4 ± 74.4	*22.4 ± 9.5	*18.2	44 ± 4
PCB 105				24.5 ± 1.1	*34.2 ± 13.5	*8.1 ± 3.2	*5.9	3.5 ± 0.6
PCB 110				63.5 ± 4.7	*120.1 ± 67.3	*29.1 ± 11.5	*18.8	
PCB 118				58.0 ± 4.3	*79.8 ± 37.1	*17.8 ± 7.7	*12.5	28 ± 3
PCB 126					*0.7			
PCB 128				8.47 ± 0.28	*14.5 ± 6.4	*4.6 ± 2.2	*2.7	5.4 ± 1.2
PCB 137					*3.8 ± 1.0	*1.7 ± 0.7		
PCB 138		1.98 ± 0.28	6.92 ± 0.52		*72.0 ± 26.3	*28.7 ± 9.7	*14.8	27 ± 4
PCB 138/163/164				62.1 ± 3.0				
PCB 141					*19.4 ± 4.0	*8.3 ± 2.0		
PCB 149				49.7 ± 1.2				49 ± 4
PCB 151		0.48 ± 0.08	1.37 ± 0.07	16.93 ± 0.36	*16.6 ± 4.9	*9.4 ± 3.7	*2.5	
PCB 153		2.27 ± 0.28	6.15 ± 0.67	74.0 ± 2.9	*68.2 ± 22.1	*27.3 ± 7.5	*11.4	50 ± 4
PCB 156				6.52 ± 0.66				3.0 ± 0.4
PCB 163								17 ± 3
PCB 169					*<0.016			
PCB 170		0.27 ± 0.05	1.07 ± 0.15		*16.8 ± 7.6	*11.8 ± 2.5	*4.3	13.4 ± 1.4
PCB 170/190				22.6 ± 1.4				
PCB 180		1.17 ± 0.15	3.70 ± 0.33	44.3 ± 1.2	*44.9 ± 23.2	*26.1 ± 11.0	*7	22 ± 2
PCB 183				12.19 ± 0.57	*15.2 ± 7.6	*8.4 ± 4.1	*1.8	
PCB 187/159/182				25.1 ± 1.0				
PCB 194		0.23 ± 0.04	0.61 ± 0.07	11.2 ± 1.4	*13.1 ± 5.6	*6.9 ± 3.1	*2.3	
PCB 195				3.75 ± 0.39				
PCB 196		0.45 ± 0.04	1.13 ± 0.12					
PCB 201		0.57 ± 0.07	1.39 ± 0.09		*7.3 ± 5.0	*8.1 ± 2.0	*0.5	
PCB 206				9.21 ± 0.51	*7.0 ± 3.0	*3.2 ± 1.6	*3.7	
PCB 209		0.33 ± 0.1	0.90 ± 0.14	6.81 ± 0.33	*1.4 ± 0.8	*1.6 ± 2.0	*9.3	
Total PCBs	1.15 ± 0.60	21.8 ± 1.1	111.8 ± 2.5		2.00 $\mu g\ g^{-1}$	*0.577 $\mu g\ g^{-1}$	*0.621 $\mu g\ g^{-1}$	

Table A4.2c. Reference materials for organic contaminants in marine sediments. Values preceded by an asterisk (*) are non-certified; all other values are certified.

Code	**SRM 1944**	**EC-1**	**EC-4**	**EC-8**	**DX-1**	**DX-2**	**BCR-462**	**EDF-2513**
Organization	**SRM-NIST**	**NWRI**	**NWRI**	**NWRI**	**NWRI**	**NWRI**	**BCR**	**CIL**
Country of origin	USA	Canada	Canada	Canada	Canada	Canada	EU	USA
Matrix	New York New Jersey Waterway sediment	Freshwater harbour sediment	Freshwater harbour sediment	Lake sediment	Great Lakes blend	Lake Ontario sediments	Coastal sediment	Fortified soil
UNITS	mg kg^{-1}	µg g^{-1}	µg g^{-1}	µg g^{-1}	pg g^{-1}	pg g^{-1}	µg kg^{-1}	ng g^{-1}
AS	Dry weight	Dry weight	Dry weight	Dry weight	Dry weight	Dry weight	Dry weight	Dry weight
[±] ***expressed as***	95% CI	± SD	± SD	± SD	95% CI	95% CI	Expanded uncertainty[1]	
UNITS OF ISSUE	50 g	100 g	100 g	100 g	50 g	50 g	25 g	10 g
FORM	Freeze-dried	Freeze-dried	Freeze-dried	Freeze-dried	Freeze-dried	Freeze-dried	Air dried	
Other chlorinated compounds	µg kg^{-1}	ng g^{-1}	ng g^{-1}	ng g^{-1}				
1,4-dichlorobenzene		*30.9		*57.65				
1,3-dichlorobenzene		*5.9	*6.8	*42.69				
1,2-dichlorobenzene		*4.9	*6.8	*5.21				
1,3,5-trichlorobenzene		*2.7	*4.4	*46.24				
1,2,4-trichlorobenzene		*3.4	*6.7	*67.01				
1,2,3-trichlorobenzene		*2.3	*1.9	*2.74				
1,2,4,5-tetrachlorobenzene		*3.4	*2.4	*56.67				
1,2,3,4-tetrachlorobenzene		*1.5	*1.6	*17.39				
1,2,3,5-tetrachlorobenzene		*0.76	*0.34	*5.53				
Pentachlorobenzene		*1.7	*1.9	*30.16				
Hexachlorobenzene		*5.4	*2.2	*97.50				
Hexachlorobutadiene		*0.66	*0.55	*21.33				
Octachlorostyrene		*6.0	*1.04	*22.18				
Antifoulants							µg kg^{-1}	
Tributyltin (TBT)							54 ± 15	
Dibutyltin (DBT)							68 ± 12	
Dioxins and furans	µg kg^{-1}	ng g^{-1}	ng g^{-1}	ng g^{-1}	pg g^{-1}	pg g^{-1}		ng g^{-1}
2,3,7,8 - TCDF	*0.039 ± 0.015				*89 ± 44	*134 ± 61		0.45 ± 0.03
1,2,3,7,8 - PCDF	*0.045 ±0.007				39 ± 14	46 ± 10		0.87 ± 0.04
2,3,4,7,8 - PCDF	*0.045 ± 0.004				62 ± 32	88 ± 28		0.86 ± 0.06
1,2,3,4,7,8 - HxCDF	*0.22 ± 0.03				714 ± 276	825 ± 348		0.88 ± 0.05
1,2,3,6,7,8 - HxCDF	*0.09 ± 0.01				116 ± 37	153 ± 61		0.95 ± 0.09
1,2,3,7,8,9 - HxCDF	*0.019 ± 0.018				*28 ± 42	*36 ± 45		0.82 ± 0.06
2,3,4,6,7,8 - HxCDF	*0.054 ± 0.006				*57 ± 36	*70 ± 47		0.91 ± 0.06
1,2,3,4,6,7,8 - HpCDF	*1.0 ± 0.1				2397 ± 796	3064 ± 745		1.27 ± 0.11
1,2,3,4,7,8,9 - HpCDF	*0.040 ± 0.006				137 ± 62	152 ± 84		1.12 ± 0.12
OCDF	*1.0 ± 0.1				7122 ± 2406	7830 ± 3087		2.25 ± 0.15
2,3,7,8 - TCDD	*0.133 ± 0.009				263 ± 53	262 ± 51		0.46 ± 0.03
1,2,3,7,8 - PCDD	*0.019 ± 0.002				22 ± 8	28 ± 14		0.96 ± 0.06
1,2,3,4,7,8 - HxCDD	*0.026 ± 0.003				23 ± 7	25 ± 8		0.90 ± 0.06
1,2,3,6,7,8 - HxCDD	*0.056 ± 0.006				77 ± 27	85 ± 33		0.87 ± 0.05
1,2,3,7,8,9 - HxCDD	*0.053 ± 0.007				53 ± 24	58 ± 19		0.90 ± 0.06
1,2,3,4,6,7,8 - HpCDD	*0.80 ±0.07				634 ± 182	757 ± 320		1.39 ± 0.10
OCDD	*5.8 ± 0.7				3932 ± 933	4402 ± 1257		3.51 ± 0.22
Total toxic equivalent (TEQ)	*0.25 ± 0.01							

[1]Expanded uncertainty $U = k.u_c$ calculated according to ISO/BIPM guide with coverage factor k = 2.

ANNEX 5

RECOMMENDED EQUATIONS FOR THE CALCULATION OF SOLUBILITY OF DISSOLVED OXYGEN IN MARINE WATERS

1 INTRODUCTION

When only physical processes are involved, the dissolved oxygen (DO) concentration in water is governed by the laws of solubility, i.e., it is a function of atmospheric pressure, water temperature, and salinity. The corresponding equilibrium concentration is generally called solubility. It is an essential reference for the interpretation of DO data. Precise solubility data, tables, and mathematical functions have been established (Carpenter, 1966; Murray and Riley, 1969; Weiss, 1970) and adopted by the international community (UNESCO, 1973). However, Weiss (1981) drew attention to an error in the international tables in which the values are low by 0.10 % since they are based on ideal gas molar volume instead of actual dioxygen molar volume. Later, the Joint Panel on Oceanographic Tables and Standards (JPOTS) recommended that the oxygen solubility equation of Benson and Krause (1984), which incorporated improved solubility measurements, be adopted and the tables updated (UNESCO, 1986). However, the UNESCO paper only referred to the equation that gives concentrations in the unit "micromole per kilogram".

The present document repeats the equations that should be used for the computation of solubility values of dissolved oxygen, in various units, according to the UNESCO recommendation. These equations (so-called B & K equations) are directly taken from the paper of Benson and Krause (1984), who provided two equations for calculation either in "micromole per kilogram" or in "micromole per litre", and the conversion factors for data in "milligram per litre" and "millilitre per litre".

2 B & K SOLUBILITY EQUATIONS

Two equations of the same type have been established for DO solubility, to obtain concentrations either in "micromole per kilogram" or in "micromole per litre". Two points should be clear:

1) in these equations, the species under consideration is dioxygen (O_2), therefore, "micromole" means "micromole of O_2";
2) 1 litre = 1 cubic decimetre, exactly.

The following symbols are used:

t : Celsius temperature (°C),
T : Kelvin temperature (K), T (K) = t (°C) + 273.15,
S : salinity on the Practical Salinity Scale 1978 (PSS78),
Cs : DO solubility concentration (the unit is mentioned using subscripts).

The equations can be expressed as follows:

$$\ln Cs_{(\mu mol\ kg^{-1})} = A + B/T + C/T^2 + D/T^3 + E/T^4 - S \times (F + G/T + H/T^2),$$

and

$$\ln Cs_{(\mu mol\ l^{-1})} = I + J/T + K/T^2 + L/T^3 + M/T^4 - S \times (N + P/T + Q/T^2).$$

The constants A to Q are the following:

	Unit
	micromole per kilogram
A =	–135.29996
B =	$+1.572288 \times 10^5$
C =	-6.637149×10^7
D =	$+1.243678 \times 10^{10}$
E =	-8.621061×10^{11}
F =	+0.020573
G =	–12.142
H =	+2363.1
	micromole per litre
I =	–135.90205
J =	$+1.575701 \times 10^5$
K =	-6.642308×10^7
L =	$+1.243800 \times 10^{10}$
M =	-8.621949×10^{11}
N =	+0.017674
P =	–10.754
Q =	+2140.7

Application domain: t = 0–40 °C; S = 0–40.

Cs is obtained as:

Cs = exp (ln Cs),

i.e., when developing the equation:

$$Cs_{(\mu mol\ kg^{-1})} = \exp [-135.29996 + (1.572288 \times 10^5) / (t + 273.15) - (6.637149 \times 10^7) / (t + 273.15)^2 + (1.243678 \times 10^{10}) / (t + 273.15)^3 - (8.621061 \times 10^{11}) / (t + 273.15)^4 - S \times (0.020573 - 12.142 / (t + 273.15) + 2363.1 / (t + 273.15)^2)].$$

and

$$Cs_{(\mu mol\ l^{-1})} = \exp[-35.90205 + (1.575701 \times 10^5)/(t + 273.15) - (6.642308 \times 10^7) / (t + 273.15)^2 + (1.243800 \times 10^{10}) / (t + 273.15)^3 - (8.621949 \times 10^{11}) / (t + 273.15)^4 - S \times (0.017674 - 10.754 / (t + 273.15) + 2140.7 / (t + 273.15)^2)].$$

3 SOLUBILITY DATA IN "MILLIGRAM PER LITRE" AND "MILLILITRE PER LITRE"

Solubility in **milligram per litre** is obtained from the value in micromole per litre by multiplying by the molar mass of dioxygen (O_2) and 10^{-3} for unit consistency, that is:

$$Cs_{(mg\ l^{-1})} = Cs_{(\mu mol\ l^{-1})} \times 0.0319988.$$

Solubility in **millilitre per litre** is obtained from the value in micromole per litre by multiplying by the molar volume of the gas at standard temperature and pressure (STP; 0 °C, 1 atmosphere). For that conversion, some data previously published refer to the molar volume (STP) of dioxygen (O_2; 0.0223916 ml per micromole), like those of Weiss (1970), while others refer to that of an ideal gas (0.022414 ml μmol^{-1}), like those of the UNESCO tables and Benson and Krause (1984). Referring to exact O_2 molar volume:

$$Cs_{(ml\ l^{-1})} = Cs_{(\mu mol\ l^{-1})} \times 0.0223916.$$

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ANNEX 6

DISEASES AND PARASITES OF BALTIC FISH

1 INTRODUCTION

The occurrence of diseases and parasites of fish in the Baltic Sea has interested scientists for many decades. However, apart from more or less anecdotal records of conspicuous phenomena such as skeletal deformities, skin lesions and grossly visible parasites, it was not before the 1970s that more systematic epidemiological studies into the prevalence, geographical distribution, and causes of diseases and parasitic infestations commenced. These studies were mainly linked to two major concerns: to a possible impact of diseases on the fishery and human consumers, and to a possible impact of anthropogenic contaminants on the health of marine fish.

At present, most Baltic Sea countries are carrying out fish disease monitoring programmes which are, however, still variable in terms of their objectives, the regional and temporal coverage as well as the target species and diseases/parasites studied (Lang and Dethlefsen, 1996). Furthermore, it is almost characteristic for monitoring programmes that the data generated are often either unpublished or scattered throughout various reports and, therefore, not readily accessible. Consequently, it is not easy to obtain a comprehensive overview of the health status of Baltic Sea fish.

This annex will, however, attempt to provide a brief description, focused on the period 1994–1998, of the prevalence and geographical distribution of selected fish diseases and parasites characteristic for the major wild Baltic Sea fish species on which most research and monitoring activities have been conducted: flounder (*Platichthys flesus*), cod (*Gadus morhua*), herring (*Clupea harengus*) and, recently, Atlantic salmon (*Salmo salar*). Other significantly affected species are also considered. In addition to the grossly visible fish diseases and parasites recommended by ICES for fish disease surveys (ICES, 1989; Bucke *et al*., 1996) and widely used for biological effects monitoring purposes, other diseases and parasites are highlighted that have received attention due to their possible link to marine contamination or their suspected impact on mortality and fish stock size. Brief information on major new trends with regard to diseases of farmed marine fish in the Baltic Sea has also been provided.

The information presented constitutes an updated version of a contribution to the HELCOM Third Periodic Assessment for the period 1989–1993 (HELCOM, 1996).

2 AVAILABLE DATA ON WILD FISH

The information provided is largely based on recent annual reports of the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO), the ACME, results from the 1994 BMB/ICES Sea-going Workshop on Fish Diseases and Parasites in the Baltic Sea, and further published or as yet unpublished information provided by Baltic Sea countries.

One of the most consistent data sets on the prevalence and spatial distribution of diseases and parasites of flounder, cod, and herring was obtained during the 1994 BMB/ICES Workshop. Practical work was conducted on board RV “Walther Herwig III” on a transect from the Mecklenburg Bight to the western Gulf of Finland, representing the largest area in the Baltic Sea ever studied for this purpose in a narrow time-window. Scientists from eight of the nine Baltic Sea countries participated in the workshop (Lang and Mellergaard, 1999).

2.1 Flounder (*Platichthys flesus*)

In addition to cod, flounder is certainly the best-studied Baltic fish species with regard to its diseases and parasites and a large body of relevant reports is available (HELCOM, 1996; Lang *et al*., 1999).

The lymphocystis disease (Plate 1.1), caused by Iridoviridae, continued to be the most prevalent externally visible disease of flounder. The prevalences observed in single sampling areas during the 1994 BMB/ICES Workshop were in the range of 4.0 % to 11.2 % (off the Estonian coast) to 22.2 % to 38.1 % (off the German coast), indicating a decreasing trend from the western to the eastern areas (Lang *et al*., 1999). Figure A6.1 (upper figure) provides information on temporal changes in prevalence for the period 1994–1999 recorded in ICES Sub-divisions 22 (inner Danish waters, the Kiel Bight, and the Mecklenburg Bight), 24 (southwestern Baltic west of Bornholm), 25 (southwestern Baltic east of Bornholm) and 26 (southeastern Baltic, including the Gulf of Gdansk), covering the southwestern Baltic Sea. A continuation of the increasing trend in prevalence reported previously for the western Baltic Sea in the period 1986–1993 (Lang and Dethlefsen, 1994) was not detected. At least for ICES Sub-division 24, the data for the period 1994–1997 indicate an opposite trend (V. Dethlefsen and T. Lang, pers. comm.).

A multivariate statistical analysis of the 1994 workshop data revealed a significant relationship between length,age, and gender of the flounder and the lymphocystis prevalence recorded. Furthermore, it could be shown that differences in these demographic factors between sampling areas to a large extent explained the differences in observed prevalence between areas (Lang *et al*., 1999). It is, therefore, important to consider these factors in any regional assessment.

The lowest prevalences of the skin ulcer disease of flounder (Plate 1.2) recorded in the course of the 1994 workshop were detected in the two western-most areas off the German coast (0.0 %). The maximum prevalence (11.6 %) occurred off the Latvian coast. The data indicated a clear increase in prevalence from the western to the eastern sampling sites (Lang *et al*., 1999). This increase is also reflected in Figure A6.1 (lower figure), which illustrates changes in prevalence over the period 1994–1998 in ICES Sub-divisions 22, 24, 25, and 26. While the data indicate an increase in prevalence for the years 1994–1996, particularly in ICES Sub-division 26, the prevalence dropped again thereafter (V. Dethlefsen and T. Lang, pers. comm.).

Kosior *et al*. (1997) observed a slight increase in prevalence from 1.05 % (1994), 1.20 % (1995) to 1.97 % (1996) at stations within the Polish Exclusive Economic Zone. These values are low compared to other data recorded in adjacent areas in the same period (Lang *et al*., 1999; V. Dethlefsen and T. Lang, pers. comm.).

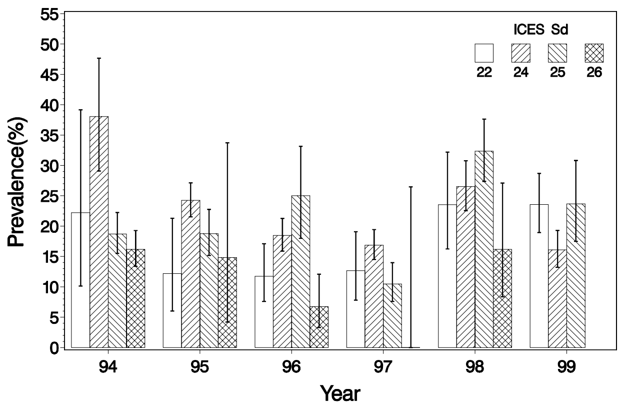
From bacteriological studies there is evidence that the skin ulcer disease in flounder is associated with one well-defined biotype of atypical *Aeromonas salmonicida*, which has also been recovered repeatedly from other Baltic fish species with acute skin ulcerations (Wiklund *et al*., 1999).

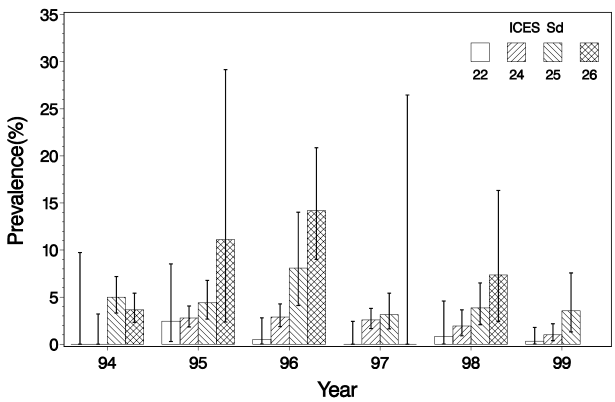
Statistical analyses have shown that skin ulcers are significantly more prevalent in male as compared to female flounder and that large fish are more often affected than small ones (Lang *et al*., 1999; Wiklund and Bylund, 1993). Furthermore, the analysis of the 1994 data strongly indicated that the disease is associated with water salinity because a negative correlation was found between the prevalence and the salinity (Lang *et al*., 1999).

The occurrence of higher prevalences of liver tumours in wild flatfish is regarded as an indicator of exposure to carcinogenic anthropogenic contaminants. Therefore, relevant studies have been incorporated in most fish disease monitoring programmes, using methodological guidelines provided by ICES (ICES, 1989; Bucke *et al*., 1996). In Baltic flounder, histologically confirmed cases of liver tumours and their putative precursors (foci of cellular alteration, FCA) have only rarely been recorded. At sampling sites in the southwestern Baltic Sea, prevalences were in the range of 0.0 %–3.0 % in the period 1994–1999 (T. Lang, pers. comm.). Slightly increased prevalences were identified in the eastern Baltic Sea off the Estonian coast (Lang *et al*., 1999; Bogovski *et al*., 1999) and at certain locations at the southwestern coast of Finland (Wiklund and Bylund, 1994b). There is no evidence for temporal trends in any of these areas.

Other gross diseases of flounder, such as acute/healing fin rot/erosion and skeletal deformities, have occurred only occasionally at generally low prevalences (< 1 %), without revealing any clear spatial or temporal trends (Lang *et al*., 1999; V. Dethlefsen and T. Lang, pers. comm.).

Figure A6.1. Prevalences (with 95 % confidence intervals) of lymphocystis (upper figure) and acute/healing skin ulcerations (lower figure) in Baltic flounder (*Platichthys flesus*) from ICES Sub-divisions 22, 24, 25, and 26 in the period 1994–1998 (after Lang *et al*., 1999; V. Dethlefsen and T. Lang, pers. comm.).





The parasitic fauna of Baltic flounder is considerably diverse, largely reflecting the changes in salinity in a northeasterly direction. A comprehensive synopsis was provided by Køie (1999) as part of the results obtained during the 1994 BMB/ICES Workshop. Typically, marine metazoan parasites (*Podocotyle atomon, Brachyphallus crenatus, Anisakis simplex, Holobomolochus confusus, Lernaeocera branchialis*) dominated in the western-most sampling areas, whereas parasites adapted to a low salinity (*Diplostomum spathaceum*, *Cotylurus* sp., *Corynosoma* sp., *Rhaphidascaris acus*, *Contracaecum osculatum*) were most common towards the east and north. Small encapsulated larvae of bird parasites (*Contracaecum* sp., *Paracuaria* sp., *Cosmocephalus* sp.) were recorded in all areas. The study confirmed earlier sporadic findings of juvenile stages of the copepod *Lernaeocera branchialis* in flounder gills, suggesting in contrast to previous reports that the whole life cycle of this parasite, the adult females of which parasitize mainly cod and other gadoids, can be completed in the Baltic Sea.

Further comprehensive information on parasites of Baltic flounder has been provided by Turovski (1994), Palm and Dobberstein (1999), and Palm *et al*. (1999). However, although the usefulness of parasites as indicators of environmental change (including effects of contaminants or eutrophication) has been suggested repeatedly, none of the reports available covering the period 1994–1998 provide information on temporal or spatial changes in the occurrence of flounder parasites or on possible links with anthropogenic impacts. However, Turovski (1994) reported a steady increase in infestation of flounder from Estonian waters with freshwater and brackish water parasites in the period 1984–1993, possibly reflecting long-term changes in salinity.

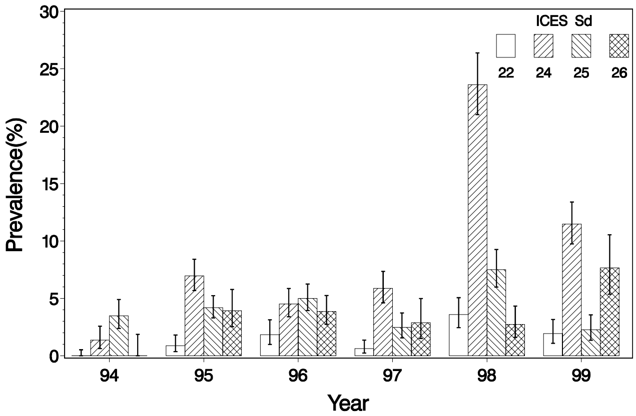
2.2 Cod (*Gadus morhua*)

Due to its commercial significance, Baltic cod has been a subject of a large variety of studies on spatial/temporal trends of externally visible diseases and parasites (see HELCOM, 1996). The most prevalent externally visible disease of Baltic cod is the infectious bacterial skin ulcer disease (Plate 1.3), often called ulcus syndrome (Jensen and Larsen, 1979).

Although there is consensus that the conspicuous disease signs (open, red ulcers affecting the skin and, occasionally, tending into the musculature) are caused by bacterial infections (e.g., *Vibrio anguillarum*, *Flavobacterium* sp., *Aeromonas* sp., *Pseudomonas* sp., often as mixed infections), the primary cause may be different. For example, Danish studies on ulcerated cod occurring at high prevalences (26 %–48 % in May) around Bornholm in 1982 revealed that these lesions were most likely caused primarily by mechanical damage to the skin of specimens which were discarded or had escaped from fishing nets (Mellergaard and Bagge, 1998) and that, subsequently, the lesions were invaded by pathogenic bacteria leading to a secondary infection. However, there is also evidence from the literature that pollution might have been involved in the aetiology of the disease in cod from the Danish Belt Sea in the late 1970s (Christensen, 1980; Jensen, 1983).

Prevalences recorded during the 1994 BMB/ICES Workshop were in the range of 0.0 %–3.8 % in relation to single sampling sites, with the highest prevalences off the Polish coast. Affected cod belonged to the size groups 22–35 cm and > 35 cm, corresponding to age groups 2 and 3, respectively, whereas smaller cod were not affected (Mellergaard and Lang, 1999). As shown in Figure A6.2, prevalences recorded in December of the years 1995–1997 in ICES Sub-divisions 22, 24, 25, and 26 were below 7 %. However, a steep increase was observed in December 1998 in ICES Sub-division 24, with a maximum prevalence of 23.4 %. A slight increase was also noted in Sub-division 25. In December 1999, the prevalence was lower, but still remained on an elevated level in ICES Sub-division 24 (V. Dethlefsen and T. Lang, pers. comm.).

Figure A6.2. Prevalences (with 95 % confidence intervals) of acute/healing skin ulcerations in Baltic cod (*Gadus morhua*) from ICES Sub-divisions 22, 24, 25, and 26 in the period 1994–1998 (after Mellergaard and Lang, 1999; V. Dethlefsen and T. Lang, pers. comm.).



In the Belt Sea, elevated prevalences were also found in May 1998 (27.6 %) and September 1999 (16.1 %) (V. Dethlefsen and T. Lang, pers. comm.) at a station in the Mecklenburg Bight. From the size distribution of the affected specimens there is evidence that the majority of diseased fish belonged to one strong year class, and it can be speculated that stock density effects might have triggered the spread of the disease.

High prevalences (maximum values at certain stations > 40 %) of skin ulcers in cod in Danish and German waters were also observed in a Danish study in spring 1998. Interestingly, subsequent virological, bacteriological, mycological, and histological studies failed to provide conclusive information on the causes of the epizootic. However, it could be confirmed that the ulcers were not caused by VHS-like virus, a speculation that had been made before, based on findings from North Sea cod (S. Mellergaard, pers. comm.). Data from sampling sites in the Polish Exclusive Economic Zone revealed a slight increase in prevalence from 0.40 % (1994) to 1.37 % (1995) and 2.76 % (1996) (Kosior *et al*., 1997).

Skeletal deformities (compression of vertebrae, lordosis and/or scoliosis, deformation of the head skeleton) (Plate 1.4) have been known for a long time to affect Baltic cod and may have multifactorial natural or anthropogenic causes (HELCOM, 1996) which most likely originate during early life stages. An impact of contaminants, in particular metals, has been discussed repeatedly. However, no clear cause-effect relationships have been established (Lang and Dethlefsen, 1987; HELCOM, 1996; Mellergaard and Lang, 1999).

From historic data, skeletal deformities in cod may occur at very high prevalences, i.e., > 50 % in the Sound and ≤ 75 % in the Gulf of Riga (Lundbeck, 1928; Berzins, 1943). During the BMB/ICES Workshop, prevalences of skeletal deformities were in the range of 0.0 %–3.8 %, according to sampling area, with the highest value in the Gulf of Gdansk. As with the skin ulcer disease, larger fish were more frequently affected than smaller ones. More recent data for the period 1995–1999 from sampling sites in the southern Baltic Proper and the Belt Sea revealed slightly higher values, with a maximum of 8.7 % in December 1998 at a station northeast of Rügen (V. Dethlefsen and T. Lang, pers. comm.). Interestingly, areas with the highest prevalences were almost identical with those characterized by elevated prevalences of skin ulcers. Prevalences recorded in Polish waters in the period 1994–1996 ranged from 0.09 %–0.39 % (Kosior *et al*., 1997).

Another disease of cod recommended for monitoring purposes (ICES, 1989; Bucke *et al*., 1996) is the X-cell disease, which leads to swelling of the pseudobranch (pseudobranchial pseudotumours) and sometimes also affects adjacent tissues (Plate 1.5). Its aetiology is still unclear, however, it has been suggested repeatedly that it is caused by parasitic protozoans (HELCOM, 1996).

During the 1994 BMB/ICES Workshop, the X-cell disease was only recorded at the two western-most stations, with a prevalence of 0.7 % and 0.2 %, respectively (Mellergaard and Lang, 1999). The authors speculate that salinity may play a role in the distribution of the disease. Monitoring data from subsequent years confirm that the disease is rare in the Baltic Sea and that it is regularly recorded only in the southwestern Baltic Sea. However, the disease was also occasionally found in more eastern waters, e.g., outside the Gulf of Gdansk (V. Dethlefsen and T. Lang, pers. comm.).

Data for the period 1994–1998 confirm earlier findings that the infestation of Baltic cod with the copepod *Lernaeocera branchialis* (Plate 1.6) in the gill chamber and encysted metacercariae of the digenean trematode *Cryptocotyle lingua* (Plate 1.7) in the skin are restricted to the southwestern part of the Baltic Sea (HELCOM, 1996). During the BMB/ICES Workshop, maximum prevalences in relation to sampling sites were 1.0 % and 21.9 %, respectively (Mellergaard and Lang, 1999). According to data from the German fish disease monitoring programme, maximum prevalences of *Lernaeocera branchialis* in the years 1995–1998 occurred in the Kiel Bight and Mecklenburg Bight and were in the range of 1.2 % to 3.6 %. For metacercariae of *Cryptocotyle lingua*, maximum values were recorded in the Kiel Bight, with prevalences ranging from 38.6 % to 62.4 % (V. Dethlefsen and T. Lang, pers. comm.). Mellergaard and Lang (1999) suggested that the pronounced spatial patterns for these parasites are determined by salinity. For *Lernaeocera*, it seems as though the parasite itself is restricted to saline waters, whereas for *Cryptocotyle*, it is likely that salinity affects the distribution of its intermediate host, the periwinkle (*Littorina littorea*).

A comprehensive checklist of parasites of Baltic cod from German waters is presented in Palm *et al*. (1999). Based on a study of ciliates in Baltic cod and flounder from the Kiel Fjord, Palm and Dobberstein (1999) discussed the usefulness of trichodinid ciliates as biological indicators of eutrophication in brackish water environments. A currently unclassified protistan endoparasite has been found in cod eggs from the Gotland Deep. Prevalences recorded were in the range of 0.0 %–75.0 % in eggs from different female fish. There was indication from incubation experiments that the infections affected the larval viability but not the embryo viability or the embryo development (Kjørsvik *et al*., 1999). Information on other studies of cod parasites has been provided in HELCOM (1996).

2.3 Herring (*Clupea harengus*)

In the Third Periodic Assessment, a description was provided of the biology, prevalence, and distribution of the two herring parasites *Anisakis simplex* and *Ichthyophonus hoferi* for the period before 1994. Both of these parasites mainly affect the migratory spring-spawning herring that invades the southern Baltic Sea in autumn/winter for spawning, coming from their feeding areas in the Kattegat, Skagerrak, and North Sea (HELCOM, 1996).

Since that time, no major change in the overall prevalence of *Anisakis simplex* has been recorded. Larval nematodes can still be found at high prevalences and intensities in the body cavity and, very rarely, in the musculature of adult herring. In the past, the highest prevalences occurred in the southwestern Baltic Sea in the major spawning areas, and they decreased towards more eastern coastal areas off the Polish coast. However, the results of recent studies indicate that the prevalence in eastern waters has increased over the past years (ICES 1998, 2000; M. Podolska, pers. comm.). During the BMB/ICES Workshop in December 1994, Grygiel (1999) detected a maximum prevalence of 39.5 % in herring from the Mecklenburg Bight and noted that only specimens larger than 20 cm were affected and that the prevalence increased with increasing body size. These findings are well in accordance with earlier findings from the same area (Lang *et al*., 1990).

The 1991 *Ichthyophonus hoferi* epizootic in herring (Plate 1.8), which caused high mortalities in the Kattegat and southwestern Baltic Sea, has declined since then and affected specimens have only rarely been found, e.g., in the Kattegat, where prevalences of 1.1 % and 0.5 % were detected in 1993 and 1998, respectively (Mellergaard and Spanggaard, 1997; ICES, 1999). However, there is evidence that the infection is enzootic to the western spring-spawning stock and that further epizootics are likely to occur. Due to the possibility of new epizootics,

Plate A6.1. Common diseases and parasites in Baltic fish species.

Plate 1.1. Lymphocystis in flounder.

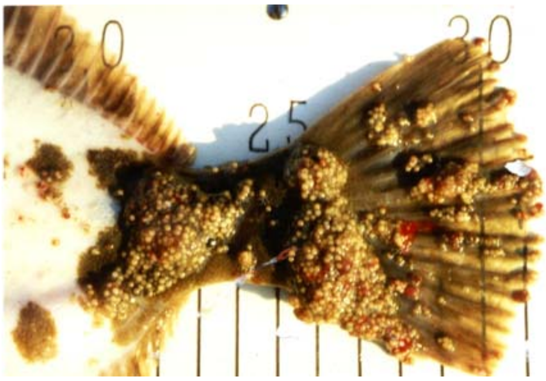


Plate 1.2. Acute skin ulceration in flounder.

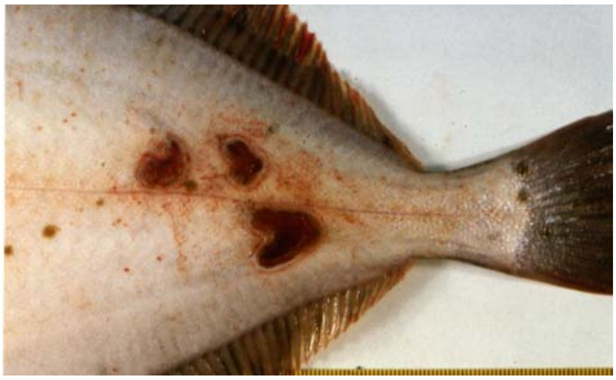


Plate 1.3. Acute skin ulceration in cod.

Plate 1.4. Skeletal deformity in cod.

Plate 1.5. X-cell disease leading to swelling in the pseudobranch in cod.

Plate 1.6. *Lernaeocera branchialis* in the gill chamber of cod.

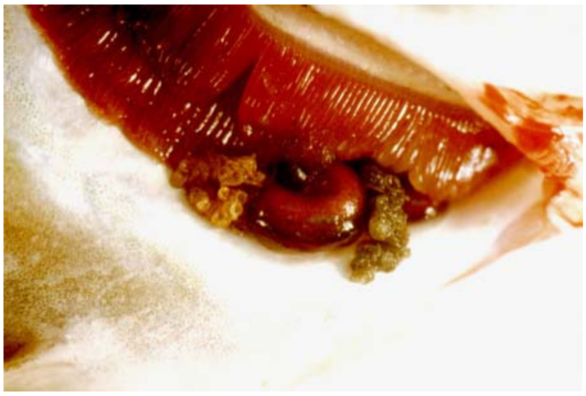
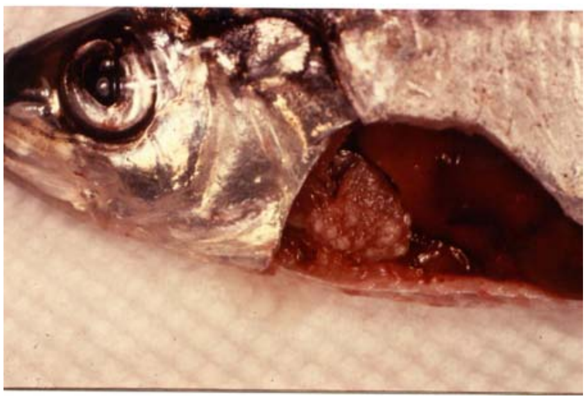


Plate 1.7. Encysted metacercariae (black spots) of *Cryptocotyle lingua* in the skin of cod.

Plate 1.8. Granulomas (white nodules) in the heart tissue of herring caused by *Ichthyophonus* sp.



ICES (1996b) has recommended that Member Countries continue to monitor the infection as part of the national stock assessment surveys in order to identify regions with potential future epizootics as early as possible and to provide useful background data for comparison purposes.

Danish studies have demonstrated the occurrence of VHS-like virus in clinically healthy herring and sprat (*Sprattus sprattus*) from the southwestern Baltic Sea. Viral Haemorrhagic Septicaemia (VHS) is one of the most important viral diseases in farmed salmonids in Europe and, originally, it was assumed that the virus was restricted to fresh water. However, over the past ten years it has been isolated from fourteen marine fish species in the North Sea and five species in the Baltic Sea (herring, sprat, cod, turbot (*Scophthalmus maximus*), and four-bearded rockling (*Enchyleopus cimbrius*)), and has also been identified in marine species in Pacific North America. Based on these findings and genetic studies, it has recently been suggested that the virus originates from the marine environment and had been introduced decades ago to the fresh water via aquaculture activities (Meyers and Winton, 1995; Dixon, 1999). Marine and freshwater viruses differ in terms of their genotype and their pathogenicity to salmonids, with the marine virus being less pathogenic. Prevalences calculated for five sampling areas visited in 1998 covering a region from the Kattegat to Bornholm were in the range of 0.0 %–4.0 % and 0.0 %–9.3 % for herring and sprat, respectively (S. Mellergaard, pers. comm.).

Other significant diseases of Baltic herring include lymphocystis, the prevalence of which recorded during the BMB/ICES Workshop ranged from 0.0 %–7.4 % depending on sampling area and, rarely, cases of skeletal deformities and epidermal hyperplasia (Grygiel, 1999).

Infection with the bacterium *Pseudomonas anguilliseptica*, associated with haemorrhagic eye lesions (Lönnström *et al*., 1994) and possibly involved in the transmission of the infection between trout mariculture facilities in Finnish waters, was still prevalent in the period 1994–1998 (G. Bylund, pers. comm.).

Studies of the parasitofauna in the Gulf of Finland and Gulf of Riga revealed a high prevalence of *Eimeria sardinae* in herring and sprat gonads, with maximum values of 80 % and 65 %, respectively, and a recently decreasing trend (Turovski *et al*., 1993; ICES, 1995a; V. Kadakas, pers. comm.). There is, however, no indication of negative effects of the parasites on affected fish.

2.4 Atlantic Salmon (*Salmo salar*)

The most spectacular health problem affecting Baltic Sea fish species is the M74 syndrome of Atlantic salmon, which continued to cause high mortalities during 1994–1998 in yolk-sac fry obtained from wild salmon and salmon artificially reared for re-stocking purposes within Swedish and Finnish compensatory stock enhancement programmes. The syndrome has been known since 1974, but massive fry mortalities have only occurred since the beginning of the 1990s (Bengtsson *et al*., 1999). Table A6.1 summarizes data recorded on salmon from different Swedish and Finnish river systems in the period 1985–1999. The highest mean prevalence (percentage of female salmon producing offspring affected by M74) recorded in female broodfish from different Swedish and Finnish rivers was 75.1 % in 1993 and fry mortalities reached levels of 90 %. After 1996, the prevalence dropped considerably, to a value of 13 % in 1998. More recent data for 1999, however, indicate another increase to 37 % (ICES, 2000).

Table A6.1. Prevalence* of the M74 syndrome in female Atlantic salmon (*Salmo salar*) used as broodfish from different Swedish and Finnish rivers (values in italics: calculation of prevalence based on less than 20 specimens) (Source of information: ICES, 2000a).

River system	ICES Sub-division	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999
Simojoki	31						*12*	0	53	74	*53*	92	*86*	86	31	38
Torne älv	31								70	74	85	66			25	56
Lule älv	31								58	66	57	48	61	38	6	34
Skellefetälven	31								40	49	69	49	77	16	4	42
Ume/Vindelälven	30	40	20	25	19	16	31	45	77	88	85	74	78	37	9	53
Ångermanälven	30								50	77	64	45	63	21	3	28
Indalsälven	30	4	7	8	7	3	8	7	45	72	65	52	64	22	1	20
Ljungan	30								*64*	*97*	*50*	56	28	*29*	10	25
Ljusnan	30							17	33	59	86	52	72	22	6	41
Dalälven	30	28	8	9	20	11	9	21	79	85	53	55	57	38	9	33
Mörrumsån	25	47	49	65	46	58	72	65	55	90	*80*	63	*56*	23		
Neva/Åland	29									70	50					
Neva/Kymi	32								45	60–70		57	*40*	79	42	
Mean total		29.8	21.0	26.8	23.0	22.0	26.4	25.8	55.8	75.1	66.4	59.1	62.0	37.4	13.3	37.0

*Prevalence: Percentage of female broodfish giving offspring with M74 symptoms. ICES Sub-divisions: 29: northern Baltic Proper and Åland Islands; 30: Bothnian Sea; 31: Bothnian Bay; 32: Gulf of Finland.

A change in the selection process of female broodfish applied as a measure to reduce M74 problems in offspring was discussed as one of the reasons for the sharp decline from 1996 to 1998. Since the occurrence of M74 can be predicted with a high degree of certainty from the condition of the adult females (behaviour, colour characteristics), apparently healthy females were selected and stripped in order to avoid fry mortality, thus leading to the observed decline in prevalence.

In Europe, M74 seems to be restricted to the Baltic Sea, where it affects salmon spawning in Swedish, Finnish, Estonian and Russian rivers, but not in Latvian rivers. There is evidence that the syndrome also affects wild Baltic sea trout (*Salmo trutta*) (Soivio, 1996; Landergren *et al*., 1999). A condition very similar to M74 is known from salmonid species in the North American Great Lakes and Finger Lakes, where the syndrome has been called Early Mortality Syndrome (EMS) or Cayuga Syndrome, respectively (Fitzsimons *et al*., 1999).

Current information from Finnish, Swedish, and North American studies provides evidence that the M74/EMS syndrome is associated with a maternally transmitted vitamin deficiency (Vitamin B_1, thiamine) (Bylund and Lerche, 1995; Bengtsson *et al*., 1999). Measurements in adults, eggs, and larvae as well as the fact that thiamine treatment of early life stages is protective against the development of M74 and EMS symptoms have provided evidence for this assumption. So far, there is no conclusive evidence on the causes of the deficiency or on the role of other potential factors involved, but dietary factors, such as changes in the abundance of the main prey species sprat and herring and their importance as food items have repeatedly been discussed as causes for M74. Furthermore, correlations have been determined between the occurrence of M74 and concentrations of dioxin-like organic contaminants which are suspected to influence the metabolism of thiamine and carotenoids. Other factors considered are eutrophication and associated algal blooms, genetic factors related to the compensatory stock enhancement programme which may lead to a genetic "bottle-neck" effect, and differences between Baltic salmon stocks in terms of migration patterns and timing of the spawning run, which may explain observed regional differences in the occurrence of the syndrome (Bengtsson *et al*., 1999; G. Bylund, pers. comm.).

2.5 Diseases in Other Fish Species

Compared to information for the period 1989–1993 (HELCOM, 1996), no major new disease outbreaks or parasitic infestations have been recorded in the Baltic Sea. Most of the diseases and parasites continued to occur without causing severe problems. However, apart from the diseases mentioned above, there are some new findings that should be followed in the future.

1) Infectious Pancreatic Necrosis (IPN)-like Birnavirus II has been isolated from dab, plaice (*Pleuronectes platessa*), cod, lemon sole (*Microstomus kitt*), and long rough dab (*Hippoglossoides platessoides*) from the Skagerrak area (S. Mellergaard, pers. comm.). However, the infections were not associated with clinical disease signs.
2) Larvae of *Anisakis* sp. were detected in garfish (*Belone belone*) from the Gulf of Finland (ICES, 2000).
3) In 10–20 % of pikeperch (*Stizostedion lucioperca*) from the Gulf of Riga, an eye pathology was observed which leads to blindness (ICES, 1999).
4) In 1995, Proliferative Kidney Disease (PKD) was recorded for the first time in wild sea trout in one of the Estonian spawning rivers. Since the disease occurred at a high prevalence (> 50 %) and was associated with clinical disease symptoms, there was concern that it might be one of the major factors leading to mortality. Since then, the causative agent of PKD, the PKX organism, has also been found in sea trout from two other rivers. From recent data, there is indication that salmon parr might also be infected (V. Kadakas, M.-L. Viilmann, and M. Kangur, pers. comm.).
5) A new digenean endoparasite, *Pseudobacciger harengulae*, was recorded in herring from the west coast of Sweden in the years 1994 to 1996 (Rahimian, 1998a). This parasite has been observed previously to affect different clupeid species in tropical and temperate regions of the Atlantic, Pacific, and Indian Oceans.
6) *Ichthyophonus hoferi* was detected in sprat and flounder from the Skagerrak and Kattegat area (Rahimian, 1998b) and in herring from the Gulf of Riga in the eastern Baltic Sea (ICES, 1996a).
7) A marked reduction in the prevalence of the parasites *Diplostomum* sp., *Glugea* sp., and *Pleistophora* sp. in flounder from the western Baltic Sea was observed in 1996 (ICES, 1997).
8) A declining trend in the occurrence of monogenean fish parasites has been observed in Estonian waters (ICES, 1996a).
9) Histopathological lesions consisting of focal hepatocellular degeneration were recorded in perch (*Perca fluviatilis*) along a polycyclic aromatic hydrocarbon (PAH) gradient at the Swedish east coast (Ericson *et al*., 1998).

Some of the other significant diseases or parasites of Baltic Sea fish species described in HELCOM (1996) continued to occur:

- High prevalences (> 50 %) of skin ulcer disease continued to occur in four-bearded rockling (*Enchelyopus cimbrius*) in deep-water regions off the

Polish coast (V. Dethlefsen and T. Lang, pers. comm.).

- Baltic cod embryos and yolk-sac larvae were found to be infested with a yet unknown protistan endoparasite (Kjørsvik *et al*., 1999; Pedersen *et al*., 1993).
- The swimbladder nematode (*Anguillicola crassus*) of the European eel (*Anguilla anguilla*), introduced to the Baltic Sea in the 1980s, continued to be widespread. However, there is indication that prevalences are lower than in pure freshwater environments and that both prevalences and intensities have tended to stabilize or even decline, possibly due to an adaptation of the host to the parasite (Svedäng, 1996). Factors limiting the distribution of the parasite seem to be high salinity and low temperature (Svedäng, 1996; Nielsen, 1997; Knopf *et al*., 1998).
- Reproductive disorders recorded in roach (*Rutilus rutilus*) in brackish waters at the Finnish coast (Wiklund and Bylund, 1994a; Wiklund *et al*., 1996), which were associated with a microsporidian infection (*Pleistophora mirandellae*), continue to be observed. This parasite has recently also been recorded in roach from Swedish brackish-water environments (Luksiene *et al*., 2000).

3 IMPACT OF ANTHROPOGENIC ACTIVITIES ON THE PREVALENCE AND GEOGRAPHICAL DISTRIBUTION OF FISH DISEASES AND PARASITES

It is now generally accepted that the identification of causes leading to changes in temporal and spatial distribution patterns of wild fish diseases is difficult. This is largely due to the fact that the aetiology of the majority of diseases is of a multifactorial nature, involving a wide range of endogenous and exogenous natural and anthropogenic stressors (Vethaak and ap Rheinallt, 1992; ICES, 1995b; Lang and Dethlefsen, 1996). However, the HELCOM Third Periodic Assessment highlighted a number of examples from the Baltic Sea for which there has been a strong indication of links between anthropogenic environmental changes and the occurrence of elevated prevalences of certain fish diseases and parasites in certain areas (HELCOM, 1996). These included effects of pulp mill effluents and discharges from other industries, temperature changes associated with discharges of heated effluents, oxygen deficiency likely to be related to eutrophication, and the introduction of non-indigenous species.

Despite improvements over the past years with regard to anthropogenic impact on the marine environmental quality of the Baltic Sea, there still exist disease problems affecting Baltic fish species which very likely are attributable to human activities:

1) The M74 syndrome continues to occur in Baltic salmon and trout and threatens the survival of the few still naturally reproducing salmon stocks in the Baltic Sea.
2) From Swedish studies there is indication that the prevalence of parasitic nematodes in perch and eelpout (*Zoarces viviparus*) might be increased in polluted areas (ICES, 2000).
3) Palm *et al*. (1999) demonstrated local differences in the occurrence of skin parasites (ciliates) of cod and flounder as indicators of eutrophication.
4) The parasitic nematode *Anguillicola crassus* introduced to Europe from Asia still persists in eel (*Anguilla anguilla*) from the Baltic Sea.
5) A new herring parasite (digenean trematode *Pseudobacciger harengulae*) has been observed at the Swedish west coast, possibly introduced via ballast water discharges (Rahimian, 1998a).
6) An elevated prevalence of liver lesions has been found in perch along a PAH gradient in Swedish coastal waters (Ericson *et al*., 1998).
7) Reproductive disorders indicating reduced reproductive capacities have been observed in female perch, roach, and pike (*Esox lucius*) from coastal Swedish and Lithuanian brackish-water areas affected by thermal effluents (Luksiene *et al*., 2000).
8) The prevalence of liver neoplasms in flounder from areas in the Gulf of Finland continues to be high (Lang *et al*., 1999; Bogovski *et al*., 1999; G. Bylund, pers. comm.) and this has been attributed to contaminants.

For other contaminant-associated diseases mentioned in the HELCOM Third Periodic Assessment (HELCOM, 1996), there is evidence that a reduction in industrial discharges has led to a decrease in disease prevalence. Examples are skeletal deformities in perch and pike and fin erosion in perch and ruffe (*Gymnocephalus cernua*) in Swedish coastal areas affected by pulp mill effluents (Lindesjöö and Thulin, 1990, 1992; Lindesjöö *et al*., 1994; E. Lindesjöö, pers. comm.).

4 IMPACT OF FISH DISEASES AND PARASITES ON BALTIC SEA FISH STOCKS

There is little conclusive information regarding the impact of fish diseases or parasites on Baltic Sea fish stocks. In only a few cases have mortalities, induced by specific diseases or environmental factors, been so obvious that it seems justified to assume that these conditions might have had a significant effect on stock size in the affected fish species (HELCOM, 1996). The *Ichthyophonus* epizootic in the beginning of the 1990s was an example of such a case. However, since it ceased shortly thereafter, it is currently not regarded as a problem for the herring stocks in the Baltic Sea. As stated above, future epizootics with possible stock effect cannot be excluded.

The M74 syndrome of Baltic salmon remains a significant problem. Although thiamine treatment of early stages of salmon and a more careful selection of adult females for breeding purposes have improved the situation in hatcheries, the syndrome undoubtedly continues to threaten the survival of the few remaining local stocks reproducing under natural conditions and, unfortunately, there is no clear indication so far of a significant and longer-lasting decrease in occurrence of the syndrome in wild salmon.

The high prevalences of acute skin ulcerations in Baltic cod observed during the past few years are causing concern that this disease might affect the stocks by significantly increasing the mortality and possibly also by reducing the fitness and reproductive potential of affected but surviving cod. Mellergaard and Bagge (1998) suggested that the ulcers they considered to be caused by mechanical damage associated with the fishery might be lethal for the fish due to the large area of the body affected. A high prevalence of skin ulcers not only potentially affects the stock due to increased mortality but also the fishery, since affected fish in most cases are not marketable due to the occurrence of conspicuous disease signs on the body surface.

There is no indication that flounder stocks are affected by the most prevalent diseases, lymphocystis and skin ulcerations. For both diseases, it is well known that skin lesions may heal and that the diseases do not affect the condition of the host in a way that it may die from the disease. Therefore, it seems unlikely that these diseases contribute significantly to natural mortality. However, again gross disease signs may affect the marketability of fish caught in a commercial fishery.

As already pointed out in HELCOM (1996), it has to be emphasized that attempts to quantify disease-induced mortalities in general suffer from a number of confounding factors and methodological problems and that, therefore, it has so far not been possible to develop and validate realistic epidemiological models for wild fish diseases, estimating disease-induced mortality rates with a high degree of probability.

5 FARMED FISH

The sea-farming activities in the Baltic Sea are mainly focused on rainbow trout (*Oncorhynchus mykiss*), with the main part of the production concentrated in the northern-most parts of the Baltic Proper, i.e., the coastal areas of Finland and Sweden. The production volume remained at a rather constant level during the 1990s, with a yearly production of 18 000 t to 22 000 t. During recent years, several new fish species have been included in the sea-farming programmes, so far more or less on a pilot scale. The most promising of these candidate species appears to be whitefish (*Coregonus lavaretus*) and char (*Salvelinus alpinus*).

Of the viral diseases, Infectious Pancreatic Necrosis (IPN) appears to be widespread in the Baltic Sea. However, rainbow trout is rather resistant to this disease and the virus is mostly recorded from fish without disease symptoms which may act as carriers. A new rhabdovirus was isolated from brown trout (*Salmo trutta*) in northern Finland in 1987 (Koski *et al*., 1992). The virus caused low mortality in brown trout fry and was characterized as European Lake Trout Virus (ELTV) (Björklund *et al*., 1994). In 1995, the virus was again isolated in brown trout without clinical symptoms in northern Finland and, in 1996, a very similar virus was isolated from brown trout in the archipelago of Stockholm. The latter fish were kept as broodfish in net cages and showed disease symptoms and low mortality. It appears that brown trout may carry the virus and remain without disease symptoms, unless stressed by handling or other rearing procedures.

Since first recorded in the Gulf of Bothnia in 1985, furunculosis caused by *Aeromonas salmonicida salm.* rapidly spread over large areas in the Baltic Sea and the Baltic Sea is now considered an endemic area for this disease. It is frequently recorded in farmed rainbow trout as well as in wild Atlantic salmon caught as broodfish. Furunculosis and vibriosis, the latter caused by *Vibrio anguillarum*, constituted the main disease threats to the Baltic sea-farming industry during the past decade. However, the significance of these two diseases has rapidly been decreasing due to the introduction of efficient vaccines and the application of large-scale vaccination programmes.

Bacterial Kidney Disease (BKD) was recorded for the first time in 1989 in the Åland Islands. Today, the disease is frequently recorded in rainbow trout sea farms in Finland and Sweden. Reports from Estonian studies in spawning rivers located in the Gulf of Finland indicate that this disease also occurs in wild sea trout (V. Kadakas, M.-L. Viilmann, and M. Kangur, pers. comm.). Farmed rainbow trout are not very sensitive to this disease and, consequently, it does not cause significant economic losses. However, in order to prevent the spread of the disease to inland waters, firm restrictions are applied in Finland and Sweden concerning movements of living fish and fish products from the Baltic Sea to fresh waters.

Flavobacteriosis caused by *Flavobacterium psychrophilum* (= *Flexibacter psychrophilus*) is another infectious disease which emerged as a serious problem in fish farms in the northern Baltic Sea during the 1990s. Although originally known as a disease in fish fry and yearlings in freshwater farms (Rainbow Trout Fry Syndrome, RTFS), infections have now frequently been recorded in rainbow trout in sea farms and appear to be an important component in the winter mortality syndrome of rainbow trout in northern fish farms.

Control of the disease is difficult since drug treatment has not been efficient and vaccines are not available so far (G. Bylund, pers. comm.).

6 CONCLUSIONS

The present compilation of available information shows that only a few new trends regarding the occurrence of diseases and parasites in wild fish from the Baltic Sea have been recorded in the period 1994–1998. However, in certain cases, significant changes in the prevalence occurred, e.g., an increase in the prevalence of skin ulcerations in cod from areas in the southwestern Baltic Sea at the end of the 1990s. A decrease in the prevalence of the M74 syndrome was observed recently. However, more current data indicate an increase in 1999 and, thus, there is still concern that M74 continues to be a major threat to the survival of the few remaining naturally reproducing salmon stocks in the Baltic Sea.

As a major new finding, the occurrence of viral infections originally thought to be restricted to farmed fish (VHS-like rhabdovirus and IPN-like Birnavirus) has been observed in wild fish. Another new finding is the occurrence of *Pseudobacciger harengulae*, a new parasite of herring from the Swedish west coast, possibly introduced via ballast water.

For a number of diseases and parasites, the impact of human activities on their prevalence and distribution remains likely. Other diseases show a clear decrease in prevalence, reflecting an improvement in the state of the environment linked to a reduction in industrial discharges. Both the negative and positive findings clearly indicate that changes in the prevalence or intensity of diseases and parasites in wild fish can be used as an indicator of biological effects of anthropogenic activities. It should, therefore, be emphasized that studies of fish diseases and parasites in wild fish in the Baltic Sea should be continued by Baltic countries and be included in future international Baltic marine monitoring and assessment programmes. Intercalibrated and standardized methodologies for fish disease surveys have been developed and established successfully, and are largely applied by national laboratories (ICES, 1989; Bucke *et al*., 1996; Lang and Mellergaard, 1999).

Regarding farmed fish, the introduction of efficient vaccines seems to largely eliminate the problems caused by the most important bacterial diseases, furunculosis and vibriosis. However, there is concern that new disease problems might emerge in the fish farming industry due to increased trade and movements of live fish and fish products between European countries and due to the introduction of new fish species in the fish farming industry.

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ANNEX 7

DISTRIBUTION AND POSSIBLE CONSUMPTION OF FISH OFFAL AND DISCARDS BY SEABIRDS IN THE BALTIC SEA

1 BREEDING POPULATIONS OF SEABIRDS IN THE BALTIC SEA

Numbers of pairs of seabirds breeding on Baltic Sea (ICES Division III) coasts were taken from Hagemeijer and Blair (1997). The data in that volume are presented by country and by distribution maps with varying sizes of symbols. As a result, there is no explicit information on the proportions of the national totals that nest on the coasts and, therefore, that may feed in the Baltic Sea rather than in freshwater or terrestrial habitats. The procedures used to convert national totals to totals for the Baltic Sea are described below. In these conversions, the following assumptions were made:

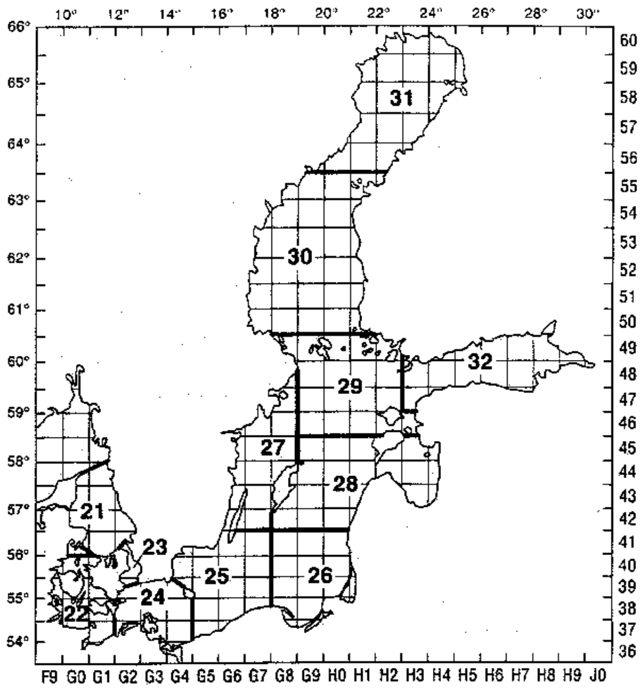
- all Arctic skuas, lesser black-backed gulls, herring gulls, great black-backed gulls, Caspian terns, Sandwich terns, common terns, Arctic terns, little terns, guillemots, razorbills, and black guillemots breeding in each country bordering the Baltic Sea fed in marine environments;
- half of the breeding cormorants, little gulls, and common gulls fed in marine environments;
- 20 % of breeding Mediterranean and black-headed gulls fed in marine environments (Table A7.1).

Table A7.1 also presents a summary of the importance and types of fish in the diet of each species using only dietary data collected from breeding colonies in the Baltic Sea, since diet composition varies between populations in different geographical regions. Similar data for wintering populations are given in Table A7.2.

It is evident from the data in Tables A7.1 and A7.2 that the terns and auks are unlikely to take discards from Baltic Sea fisheries as they feed predominantly on very small fish, with a tendency to select lipid-rich fish such as clupeids. There is only slight evidence from existing dietary studies for breeding large gulls to take discards as a part of their breeding season diet, but this has not been studied in detail in the Baltic area and there have been few recent studies of gull diet, and none specifically investigating the occurrence of discards in breeding Baltic gull diets. In winter, it seems likely that herring gulls will predominate as scavengers of discards in the Baltic given their numerical abundance (Table A7.2), with fulmars, common gulls, and kittiwakes competing for offal and taking small discards, and great black-backed gulls taking large fish discards. Seaducks are extremely abundant in the Baltic in winter, and there is some evidence to indicate that they can deplete their bivalve food resources through the winter and that they switch increasingly to feeding on fish (especially sandeels) and fish eggs during spring (Stempniewicz and Meissner, 1999). Eider ducks do feed on discarded fish and offal beside small fishing boats in harbours, and can become very tame when regularly given fish scraps (R.W. Furness, pers. comm.). It is possible that seaducks, particularly eiders, may consume some discards and offal from Baltic fisheries, but this has not been reported or investigated. The ability of eiders to dive (though rarely to depths greater than 30 m) could permit them access to sinking discards and offal out of the reach of surface-feeding gulls and fulmars.

In the case of the herring gull in the Baltic during winter, Durinck *et al.* (1994) stated that "more than half occur in the Bornholm Deep, where intensive fishing activities are carried out". Most of the other herring gulls are found in Sub-divisions 21–23 (Figure A7.1), corresponding to the areas identified by the Study Group on Estimation of the Annual Amount of Discards and Fish Offal in the Baltic Sea (SGDIB) as the Sub-divisions where the vast majority of discards and the majority of offal are produced.

Figure A7.1. Map of the ICES Fisheries Divisions and Sub-divisions of the Sound, Belt Sea, and Baltic Sea used in fishery assessment.



Division IIIa includes Sub-division 21 (plus Skagerrak)
Division IIIb is equivalent to Sub-division 23
Division IIIc is equivalent to Sub-division 22
Division IIId is divided into Sub-divisions 24–32

Table A7.1. Breeding numbers of seabirds in Baltic Sea countries, estimates of the numbers of pairs of breeding seabirds that feed in Baltic Sea marine environments during summer, and summary of importance of fish in the diet (Hagemeijer and Blair, 1997).

Seabird species	Breeding pairs in Baltic countries	Breeding pairs feeding in Baltic Sea in summer	Fish in breeding season diet of Baltic Sea colonies	References for diet composition data
Great cormorant	60,000	30,000	100 % fish, of a wide range of species and sizes	Cramp and Simmons, 1977
Arctic skua	1,000	1,000	Mostly small fish stolen from terns and gulls	Cramp and Simmons, 1983
Mediterranean gull	50	10	Some marine fish including discards, but mainly outside breeding season	Cramp and Simmons, 1983
Little gull	13,000	6,500	Mostly insects, but some fish	Cramp and Simmons, 1983
Black-headed gull	600,000	120,000	14 % fish	Götmark, 1984
Common gull	100,000	50,000	17–49 % fish	Götmark, 1984
Lesser black-backed gull	30,000	30,000	87–92 % fish (herring, sandeels, plaice, gadoids), including discards up to 31 cm in length	Götmark, 1984; Hario, 1990
Herring gull	200,000	200,000	21–60 % fish, including discards	Götmark, 1984
Great black-backed gull	12,000	12,000	41–82 % fish, including discards	Götmark, 1984
Kittiwake	100	100	Mostly sprat	Cramp, 1985
Caspian tern	1,900	1,900	Fish, especially roach, perch and clupeids, 15–20 cm	Cramp, 1985
Sandwich tern	1,500	1,500	Marine fish, ~12 cm	Cramp, 1985
Common tern	70,000	70,000	Small marine and brackish fish, ~6 cm	Cramp, 1985
Arctic tern	80,000	80,000	Small marine and brackish fish, ~6 cm	Cramp, 1985
Little tern	2,000	2,000	Marine invertebrates and small fish, ~4 cm	Cramp, 1985
Common guillemot	15,000	15,000	Mostly sprat	Lyngs and Durinck, 1998
Razorbill	15,000	15,000	Mostly clupeids	Cramp, 1985
Black guillemot	20,000	20,000	Littoral and sublittoral fish and crustacea, especially butterfish, sandeels, and blennies	Cramp, 1985

Table A7.2. Numbers of seabirds in Baltic Sea marine environments outside of the breeding season, with a summary of the importance of fish in the diet, and assessment of the likely importance of discards and/or offal in their diet.

Seabird species	Number of individuals in winter	Fish in diet outside of the breeding period, using Baltic Sea data where possible	Likely importance of discards/offal in diet outside breeding season	References for diet composition data relating to non-breeding seasons
Great cormorant	19,400	100 % fish, of a wide range of species and sizes	Probably not, though there are a few records of cormorants taking discards.	Cramp and Simmons, 1977
Northern fulmar	2,700	No Baltic data? Diet in Atlantic and North Sea includes offal and discards, as well as zooplankton and small fish	Yes. Fulmars often obtain high proportions of offal and some discards.	Cramp and Simmons, 1977; Tasker and Furness, 1996
Seaducks (mostly eiders, scoters, long-tailed ducks)	7,500,000	0–40 % fish (mainly sandeels and fish eggs), fish consumption increasing from winter to spring	Probably not, though seaducks, especially eiders, might take some discards and offal.	Stempniewicz and Meissner, 1999
Little gull	2,250	Probably chiefly fish and marine invertebrates	Not known. Due to small size may not be able to compete with larger gulls.	Cramp and Simmons, 1983
Common gull	72,000	Some fish, including clupeids	Yes? However, may be outcompeted by larger gulls.	Cramp and Simmons, 1983
Herring gull	310,000	5–30 % gadoid discards; highest in winter and higher off Rostock than off Kiel	Yes. Takes discards and, where possible, offal.	Garthe *et al.*, unpubl. data; Garthe and Scherp, unpubl. data; Scherp, 1999
Great black-backed gull	21,000	Mainly fish	Yes. Takes discards and can swallow fish up to 35 cm length.	Cramp and Simmons, 1983
Kittiwake	76,000	Mainly marine fish and invertebrates	Yes. However, may be outcompeted by larger gulls.	Cramp and Simmons, 1983
Common guillemot	86,000	46 % clupeids (mean length 13 cm), 51 % gadoids (mean length = 10 cm, max. = 21 cm)	No.	Lorentsen and Anker-Nilssen, 1999
Razorbill	156,000	Similar to common guillemot	No.	Cramp, 1985
Black guillemot	50,000	Marine inshore fish	No.	Cramp, 1985

According to Durinck *et al.* (1994), “outside the breeding season, herring gulls frequently eat fish waste from trawlers, and their winter distribution in the offshore parts of the Baltic Sea seems to depend on commercial fishing activities”. Total counts of wintering herring gulls in the Baltic do not include birds in harbours, on coastlines, in estuaries or in terrestrial habitats, although birds may move between these habitats. Durinck *et al.* (1994) stated of other gulls that might feed on discards or offal, “mew gulls [= common gulls] may also feed on fisheries waste”, of great black-backed gulls “this species is a common scavenger behind trawlers, where it feeds mainly on discarded fish” and of kittiwakes “fish waste from fishing vessels is also an important food resource”.

It can therefore be surmised that these gulls are important scavengers of discards and offal from the Baltic Sea fisheries. Next, a more quantitative assessment of this feeding activity is developed.

2 QUANTITIES OF DISCARDS AND OFFAL PRODUCED BY BALTIC FISHERIES

The Study Group on Estimation of the Annual Amount of Discards and Fish Offal in the Baltic Sea (SGDIB) concluded that in 1998 (the year for which the best data were available) about 11 000 t of fish were discarded in the Baltic, with 4432 t in Sub-division 22, 2058 t in Sub-division 24, 3598 t in Sub-division 25, 786 t in Sub-division 26, and only 125 t in all the Sub-divisions of the Baltic further to the north (Sub-divisions 27–32). No data on sizes of fish discarded were available, but the species composition was predominantly cod (6573 t), flounder (2089 t), sprat (910 t), plaice (515 t), dab (390 t), scorpion fish (217 t), turbot (68 t), and whiting (52 t). The total discard mass represented 1.4 % of the total Baltic fish catch in 1998, or 3.8 % of the catch if the industrial fish catch is excluded. This is a remarkably low discard fraction by comparison with fisheries in the North Sea and other areas. The estimate of offal discharged at sea was 15 000 t, mostly discharged in Sub-divisions 22–26, with similar quantities produced in the first, second, and fourth quarters, and about half in the third quarter of the year.

3 ESTIMATES OF FISH CONSUMPTION BY SEABIRDS IN THE BALTIC AREA

The fish consumption by seabirds of the species expected to feed extensively on discards or offal was estimated as shown in Table A7.3. The result suggests that the seabirds likely to feed on discards and offal have a total fish consumption in the Baltic Sea that is considerably greater than the estimated quantity of discards and offal being produced there. This is not necessarily a contradiction, since it is known that several of these seabird species catch fish such as herring, sprat, and sandeels (Tables A7.1 and A7.2) as well as feeding on discards and offal.

Table A7.3. Estimates of the quantities of fish (including discards and offal) consumed in the Baltic Sea by seabirds of the species likely to take some discards and/or offal.

Seabird species	Estimated fish consumption (tonnes)	Time of year of highest consumption
Northern fulmar	64	Winter
Black-headed gull	1 867	All year
Common gull	3 516	All year
Herring gull	29 923	Winter
Lesser black-backed gull	4 944	Summer
Great black-backed gull	4 557	Winter
Black-legged kittiwake	977	Winter
Total	45 848	Mainly in winter

3.1 Consumption of Discards and Offal by Seabirds at Fishing Vessels in the Southern Baltic

Discard and offal consumption was investigated by so-called discard experiments (Tables A7.4, A7.5, and A7.6) from commercial fish trawlers and fishery research vessels (Scherp, 1999). The study was conducted between June and December 1998, following the general methodology described, e.g., by Hudson and Furness (1988) and Camphuysen *et al.* (1995). Studies concentrated on the areas off Kiel and Rostock, but also took place up to Polish waters. Table A7.4 shows that the proportions of experimentally discarded fish from commercial and research vessels that were consumed by gulls were high for roundfish, but lower for flatfish. For roundfish, over 80 % of the discards up to lengths of 27 cm were taken by gulls, but larger cod were often too big for gulls to swallow. By species of fish (Table A7.5), consumption rates by gulls were similar to results obtained previously in studies in the North Sea (Camphuysen *et al.*, 1995). Consumption was predominantly by herring gulls, which took over 90 % of the discards consumed by seabirds, both in studies in ICES Division IIIc and in Division IIId (Table A7.6). These data are based on relatively small numbers of cruises and there is a need for further studies of discard consumption by seabirds in the Baltic, but the patterns reported here are as would be anticipated and are unlikely to be altered by more detailed study.

3.1.1 Ship-following seabirds

Birds attending commercial trawlers and fishery research vessels were counted throughout the study period. Following Garthe and Hüppop (1994), the maximum number per bird species attracted by the fishing vessels was recorded. From that, mean numbers of ship-followers as well as their relative proportion were calculated (Table A7.7). Herring gulls predominated behind fishing vessels, forming 90–96 % of the total flock of scavenging seabirds in the Baltic, both in summer and in winter. These results are entirely

Table A7.4. Proportion of discarded fish (by length) consumed by scavenging seabirds in the Baltic Sea. ICES Divisions IIIc and IIId are combined as are seasons. Data are calculated from Scherp (1999) and Garthe and Scherp (unpubl. data). PDC = proportion of discards consumed.

Length (cm)	Cod sample	PDC	Dab sample	PDC	Sprat sample	PDC	Herring sample	PDC	Whiting sample	PDC
7	1	0	0	-	0	-	0	-	0	-
8	0	-	0	-	5	60	1	0	0	-
9	1	100	0	-	4	100	0	-	3	100
10	8	75	4	50	27	93	1	0	12	100
11	5	80	4	50	203	99	2	100	16	94
12	23	87	5	80	274	98	7	71	23	87
13	19	89	5	0	90	96	10	100	40	80
14	26	96	4	25	36	94	7	86	60	93
15	31	100	4	50	0	-	15	93	91	98
16	26	96	4	0	0	-	59	97	79	92
17	33	94	10	20	0	-	67	97	40	98
18	33	85	11	27	0	-	50	96	24	88
19	41	95	6	0	0	-	35	97	22	100
20	81	95	28	21	0	-	3	100	37	86
21	75	92	31	3	0	-	0	-	14	100
22	89	84	16	25	0	-	0	-	27	93
23	88	95	19	16	0	-	0	-	14	100
24	93	86	9	44	0	-	0	-	7	100
25	171	83	14	29	0	-	0	-	19	89
26	78	83	11	36	0	-	0	-	8	88
27	60	73	4	25	0	-	1	100	8	88
28	57	72	0	-	0	-	0	-	5	100
29	46	67	1	0	0	-	0	-	5	60
30	40	65	1	100	0	-	0	-	4	100
31	18	39	0	-	0	-	0	-	0	-
32	21	48	0	-	0	-	0	-	0	-
33	10	70	0	-	0	-	0	-	0	-
34	9	22	0	-	0	-	0	-	1	100
35	5	20	0	-	0	-	0	-	1	0
36	5	20	0	-	0	-	0	-	0	-
37	0	-	0	-	0	-	0	-	0	-
38	2	50	0	-	0	-	0	-	0	-
39	2	100	0	-	0	-	0	-	0	-
40	1	0	0	-	0	-	0	-	0	-
41	0	-	0	-	0	-	0	-	0	-
42	0	-	0	-	0	-	0	-	0	-
43	0	-	0	-	0	-	0	-	0	-
44	0	-	0	-	0	-	0	-	0	-
45	0	-	0	-	0	-	0	-	0	-
46	2	0	0	-	0	-	0	-	0	-

Table A7.5. Consumption of different discard species by seabirds. Calculations are given for ICES Divisions IIIc and IIId, but seasons are combined due to sample sizes. Data are calculated from Scherp (1999) and Garthe and Scherp (unpubl. data).

Species discarded	ICES Division IIIc		ICES Division IIId	
	Sample	Proportion consumed	Sample	Proportion consumed
Cod	1119	85 %	1183	84 %
Dab	191	23 %	-	-
Herring	46	78 %	828	89 %
Plaice	11	0 %	-	-
Four-bearded rockling	16	94 %	18	61 %
Sprat	15	73 %	935	89 %
Scad	6	83 %	-	-
Whiting	580	93 %	40	83 %

Table A7.6. Consumption of three types of discards by seabirds. Calculations are given for ICES Divisions IIIc and IIId, but seasons are combined due to sample sizes. Data are calculated from Scherp (1999) and Garthe and Scherp (unpubl. data).

	Sample size	Proportion consumed by species				
		Herring gull	Great black-backed gull	Lesser black-backed gull	Common gull	Black-headed gull
ICES Division IIIc						
gadids	1485	95 %	5 %	0 %	0 %	0 %
clupeids	47	91 %	0 %	0 %	9 %	0 %
flatfish	44	91 %	9 %	0 %	0 %	0 %
ICES Division IIId						
gadids	1023	93 %	6 %	0 %	1 %	0 %
clupeids	1573	96 %	0 %	2 %	1 %	0 %
flatfish	-					

Table A7.7. Absolute and relative proportions of the seabirds attending fishing vessels in the Baltic Sea. Data are calculated from Scherp (1999) and Garthe and Scherp (unpubl. data). Great cormorant (*Phalacrocorax carbo*) and little gull (*Larus minutus*) were seen once and are not listed here.

	Sample size (No. of counts)	Species attending				
		Herring gull	Great black-backed gull	Lesser black-backed gull	Common gull	Black-headed gull
Summer						
ICES Division III	11	31 (96 %)	2 (3 %)	0	<1 (1 %)	0
ICES Division IIId	12	129 (91 %)	4 (3 %)	7 (5 %)	1 (1 %)	1 (1 %)
Winter						
ICES Division IIIc	18	117 (91 %)	5 (4 %)	0	6 (5 %)	< 1 (0 %)
ICES Division IIId	22	249 (90 %)	13 (5 %)	0	16 (6 %)	0

consistent with expectations based on the total numbers nd the feeding habits of the seabirds to be found in the Baltic (Tables A7.1 and A7.2).

3.1.2 Estimated quantities of discards and offal consumed by Baltic seabirds

Quantities of discards and offal consumed by seabirds in the Baltic area (Table A7.8) were calculated on the basis of the estimates of discards and offal production provided by SGDIB, as summarized in Section 11 of this report. These values were multiplied by the relative proportions of discards and offal consumed by seabirds so that a total of 6900 t of discards and 14 500 t of offal resulted. According to Section 11, discards represent only 3.8 % of the total catch (with industrial catch totals excluded), which is remarkably low compared to all studies in the North Sea (e.g., Garthe *et al.*, 1996). The discard and offal utilization by birds, on the other hand, might be somewhat overestimated due to a bias by conducting the discard experiments (Garthe and Hüppop, 1998). Further studies and the application of correction figures (Garthe and Hüppop, 1998) are certainly needed to refine the estimates on discard and offal consumption by seabirds, but it can be seen that gulls take a very high proportion of the discards and offal produced by fisheries in the Baltic Sea, with the herring gull being the main consumer at all times of the year.

Table A7.8. Quantities of discards and offal discharged annually in the Baltic Sea and their consumption by seabirds. Data on discard and offal quantities were taken from Section 11. Proportions consumed by seabirds were calculated as follows: for cod, sprat, and whiting, values were taken from Table A7.5; for flatfish, values were taken as an average for dab and plaice (see Table A7.5); for scorpion fish and "other fish", consumption percentages were set at 50 %.

Species	Amount (tonnes)	Proportion consumed by seabirds	Amount consumed by seabirds (tonnes)
Cod	6,573	85 %	5,587
Flounder	2,089	12 %	251
Sprat	910	81 %	737
Plaice	515	12 %	62
Dab	390	12 %	47
Scorpion fish	217	50 %	109
Turbot	68	12 %	8
Whiting	52	88 %	46
Other fish	~188	50 %	94
Total discards	**11,002**	**63 %**	**6,941**
Total offal	**14,950**	**97 %**	**14,502**

4 CONCLUSIONS

Although there has been only a very limited amount of work on the consumption of discards by seabirds in the Baltic Sea, it is evident that herring gulls consume a high proportion of the offal and discard production in this area. A few discards are too large for gulls to swallow (predominantly cod over 27 cm), but there is evidence of gulls selecting roundfish discards and not taking some flatfish discards. The data suggest that gulls consume considerably more than half of the discards and almost all of the offal discharged by Baltic fisheries. In addition, it appears that herring gull distribution in the Baltic in winter is determined to a considerable extent by the local distribution of discarding fishing vessels, which are concentrated in the southwestern Baltic. Whether the provision of discards and offal from Baltic fisheries has affected the population trends of gulls in the Baltic is not known.

5 ACKNOWLEDGEMENT

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ANNEX 8

MAPPING OF FISH AND SHELLFISH DISEASES IN ICES MEMBER COUNTRIES

The purpose of mapping the spatial distributions and temporal trends of fish and shellfish diseases is to give people, e.g., scientists, managers, laypersons and politicians, having interest in or needing information on this field the possibility of obtaining a rapid overview. The work was carried out through the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) and the material is based on national reports provided by the WGPDMO members. The data were processed by W. Wosniok (Germany), T. Lang (Germany), and S. Mellergaard (Denmark).

DISCLAIMER – WARNING

It should be noted that these illustrations of the spatial distribution and the temporal trends are gross overviews and will not allow for detailed interpretations on a local scale.

The mollusc diseases presented on the following maps were selected because they constitute some of the most important diseases in oyster culture.

The maps showing the distribution of marine Viral Haemorrhagic Septicaemia (VHS)-like virus maps were presented because this virus may constitute a threat to marine aquaculture. In future, it is the intention to further expand these illustrations to include other diseases of fish and shellfish.

Diseases of Dab (*Limanda limanda*): ICES Data Inventory

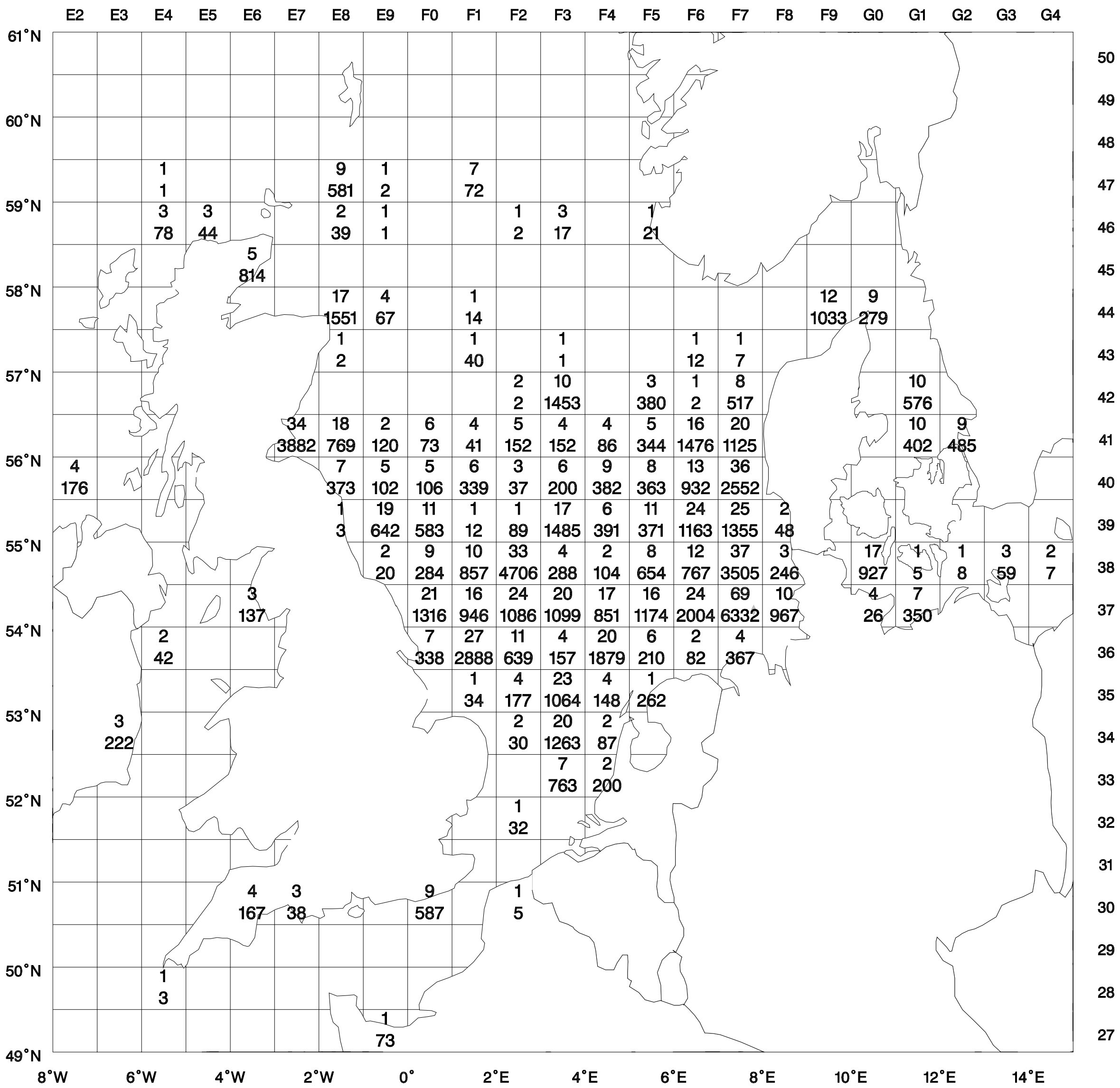

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Figure A8.1. ICES Data Inventory.

Background. Data used for the statistical analysis of trends in the prevalence of externally visible diseases of the common dab (*Limanda limanda*) have been extracted from the fish disease database of the ICES Environmental Data Centre. The fish disease database consists of data on the prevalence of diseases in wild fish submitted by ICES Member Countries conducting fish disease monitoring programmes.

All steps involved in the practical work during fish disease surveys (sampling strategies, inspection of fish for target diseases, disease diagnosis) as well as reporting and validation of data submitted to the ICES Environmental Data Centre are done according to ICES standard quality assurance procedures. Coordination of these activities is within the long-term remit of the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO).

Content of Figure A8.1. For each ICES statistical rectangle (0.5° latitude, 1.0° longitude), the map provides information on the number of observations (sampling dates) in the period 1981–1997 (upper number) and on the number of fish examined and incorporated in the statistical analysis (lower number).

In order to enable a regional comparison of trends, information is only presented for female dab of the size group 20–24 cm total length. These criteria have been selected because:

- *fish of this size are generally abundant in the study area;*
- *variation in age in this size group is smaller than in larger fish;*
- *female fish are more abundant than male fish;*
- *sex-specific variations in disease prevalence occur and data for females and males should, therefore, not be combined;*
- *the selection criteria correspond to those applied in other monitoring programmes, e.g., chemical monitoring within the OSPAR Coordinated Environmental Monitoring Programme (CEMP).*

Conclusions from Figure A8.1. Data on the occurrence of externally visible diseases in dab (females, size group 20–24 cm) are available for many ICES statistical rectangles. However, the number of observations (sampling dates) and the number of fish examined differ considerably between rectangles. That means that temporal trends cannot be calculated for all of these rectangles.

Diseases of Dab (*Limanda limanda*): Trends 1993 – 1997 for Lymphocystis

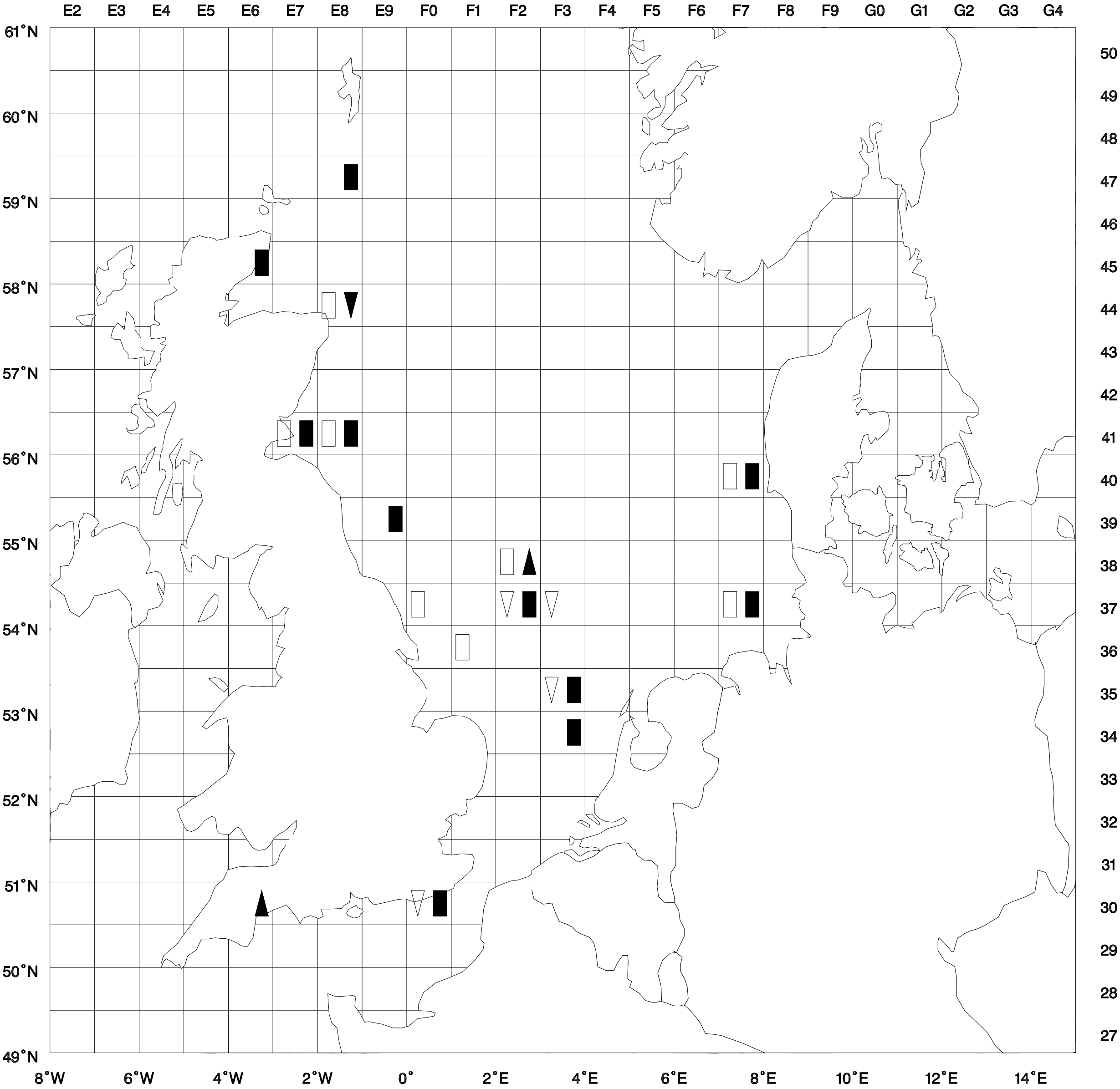

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Figure A8.2. Temporal trends in the prevalence of lymphocystis in common dab (*Limanda limanda*) in the period 1993–1997.

Background. Since the intention of the ICES Environmental Status Report is to provide current information on the quality of the marine environment, Figure A8.2 shows trends in the prevalence of lymphocystis in dab for the period 1993–1997, calculated from the most recent data available in the ICES Environmental Data Centre.

Trends have been identified using statistical procedures based on logistic regression analysis. More details on the method are provided elsewhere.

Content of Figure A8.2. Current temporal trends for the estimated prevalence of lymphocystis in female dab, size group 20–24 cm (see Figure A8.1 for sampling information), are presented as upward or downward arrows or as rectangles, representing respectively significantly increasing, decreasing, or "stable" prevalences in the period 1993–1997.

Since seasonal effects on the prevalence have been observed and bias due to combining data from different seasons should be avoided, data are shown for two seasons:

Season 1: April–September, filled symbols;

Season 2: October–March, empty symbols.

Trends are shown only for those rectangles for which the data available meet the following criteria:

- prevalence data must be available for a minimum of four out of the five years considered;
- these data must originate either from Season 1 or Season 2 or from both.

Conclusions from Figure A8.2. Only for nine rectangles were sufficient data available for trend calculation for both seasons. With the exception of only two rectangles (30E6, 38F2), prevalences of lymphocystis were either decreasing or did not show a trend in the period 1993–1997.

Diseases of Dab (*Limanda limanda*): Trends 1993 – 1997 for Epidermal Hyperplasia/Papilloma

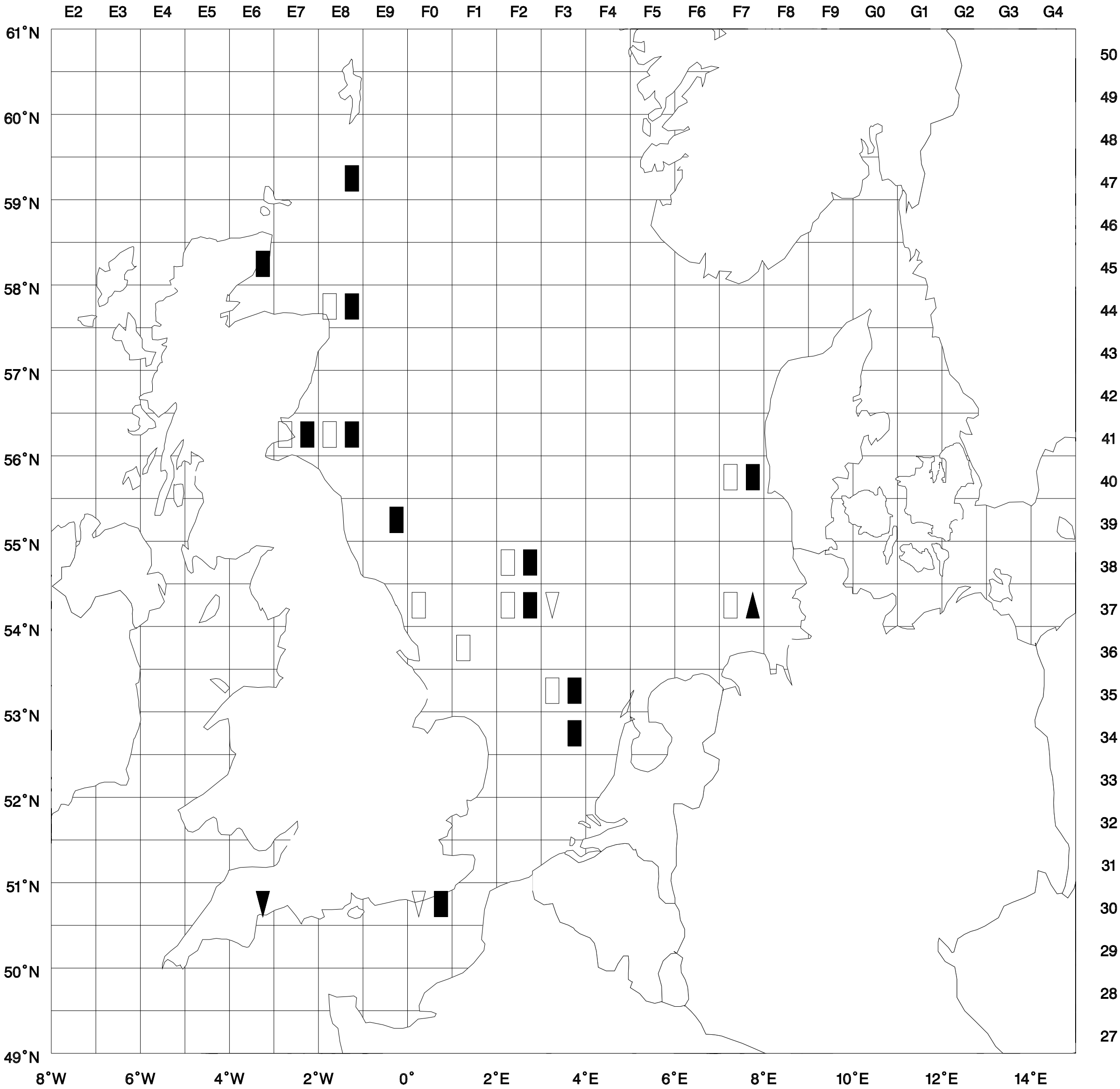

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Figure A8.3. Temporal trends in the prevalence of epidermal hyperplasia/papilloma in common dab (*Limanda limanda*) in the period 1993–1997.

Background. Since the intention of the ICES Environmental Status Report is to provide current information on the quality of the marine environment, Figure A8.3 shows trends in the prevalence of epidermal hyperplasia/papilloma in dab for the period 1993–1997, calculated from the most recent data available in the ICES Environmental Data Centre.

Trends have been identified using statistical procedures based on logistic regression analysis. More details on the method are provided elsewhere.

Content of Figure A8.3. Current temporal trends for the estimated prevalence of epidermal hyperplasia/papilloma in female dab, size group 20–24 cm (see Figure A8.1 for sampling information), are presented as upward or downward arrows or as rectangles, representing respectively significantly increasing, decreasing, or “stable” prevalences in the period 1993–1997.

Since seasonal effects on the prevalence have been observed and bias due to combining data from different seasons should be avoided, data are shown for two seasons:

Season 1: April–September, filled symbols;

Season 2: October–March, empty symbols.

Trends are shown only for those rectangles for which the data available meet the following criteria:

- prevalence data must be available for a minimum of four out of the five years considered;
- these data must originate either from Season 1 or Season 2 or from both.

Conclusions from Figure A8.3. Only for nine rectangles were sufficient data available for trend calculation for both seasons. With the exception of only one rectangle (37F7), prevalences of epidermal hyperplasia/papilloma were either decreasing or did not show a trend in the period 1993–1997.

Diseases of Dab (*Limanda limanda*): Trends 1993–1997 for Acute/Healing Skin Ulcerations

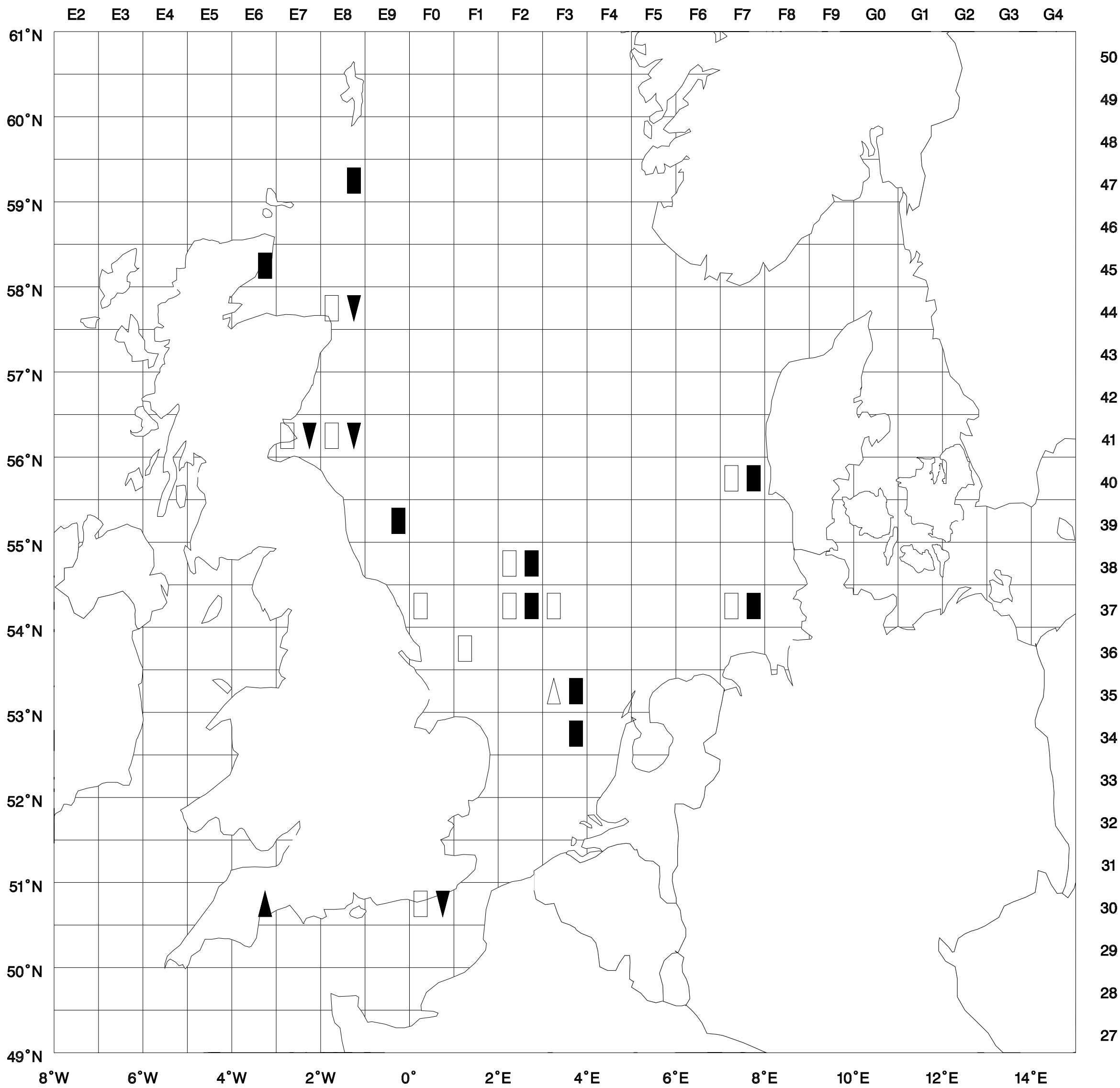

Figure A8.4. Temporal trends in the prevalence of acute/healing skin ulcerations in common dab (*Limanda limanda*) in the period 1993–1997.

Background. Since the intention of the ICES Environmental Status Report is to provide current information on the quality of the marine environment, Figure A8.4 shows trends in the prevalence of acute/healing skin ulcerations in dab for the period 1993–1997, calculated from the most recent data available in the ICES Environmental Data Centre.

Trends have been identified using statistical procedures based on logistic regression analysis. More details on the method are provided elsewhere.

Content of Figure A8.4. Current temporal trends for the estimated prevalence of acute/healing skin ulcerations in female dab, size group 20–24 cm (see Figure A8.1 for sampling information), are presented as upward or downward arrows or as rectangles, representing respectively significantly increasing, decreasing, or "stable" prevalences in the period 1993–1997.

Since seasonal effects on the prevalence have been observed and bias due to combining data from different seasons should be avoided, data are shown for two seasons:

Season 1: April–September, filled symbols;

Season 2: October–March, empty symbols.

Trends are shown only for those rectangles for which the data available meet the following criteria:

- prevalence data must be available for a minimum of four out of the five years considered;
- these data must originate either from Season 1 or Season 2 or from both.

Conclusions from Figure A8.4. Only for nine rectangles were sufficient data available for trend calculation for both seasons. With the exception of only two rectangles (30E6, 35F3), prevalences of acute/healing skin ulcerations were either decreasing or did not show a trend in the period 1993–1997.

The geographical distribution of *Bonamia ostreae*.

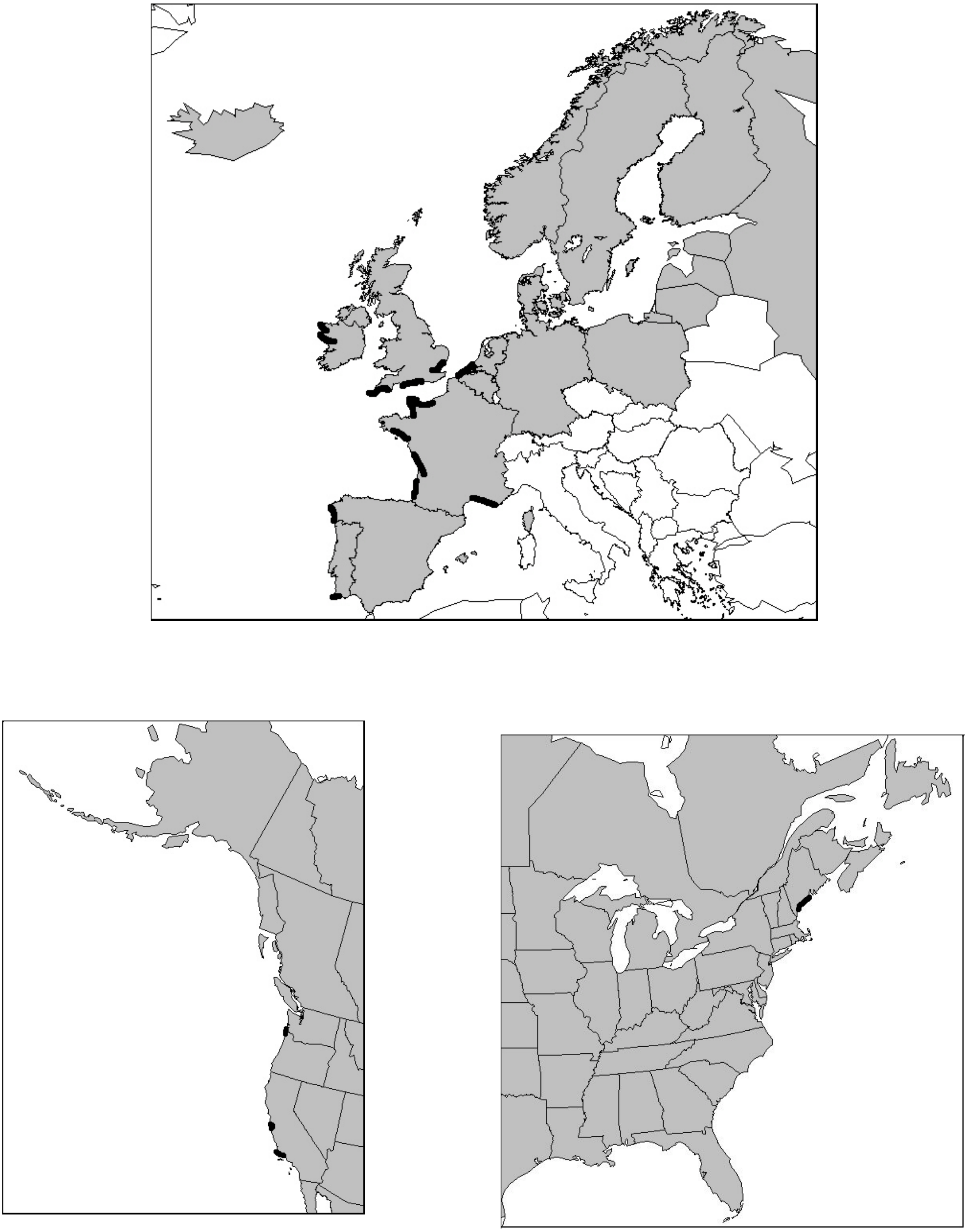

Figure A8.5. The geographical distribution of *Bonamia ostreae.*

Introduction. *Bonamia ostreae* is an intracellular parasite (2–4 μm) affecting the haemocytes of the flat oyster, *Ostrea edulis*. The parasite was observed for the first time in Normandy, France in 1979 and has since spread to other European countries associated with transfers of oysters. In affected areas, it is not economically feasible to produce flat oysters due to high mortalities (40–60 %). In the USA, the *Bonamia* situation seems to be at an endemic level without causing significant mortality.

Although many infected oysters appear normal, others may have yellow discolouration and/or extensive lesions (i.e., perforated ulcers) on the gills and mantle. Actual pathology appears correlated to haemocyte destruction and haemocytic infiltration of the connective tissue due to proliferation of *B. ostreae*. Lesions occur in the connective tissues of the gills, mantle, and digestive gland. Although some flat oysters die with light infections, others succumb to much heavier infections. Heavily infected oysters tend to be in poorer condition than uninfected oysters.

Content of the map. The map illustrates the geographical distribution of *Bonamia ostreae* (heavy black line).

Source of the data. WGPDMO members.

The geographical distribution of *Marteilia refringens*.

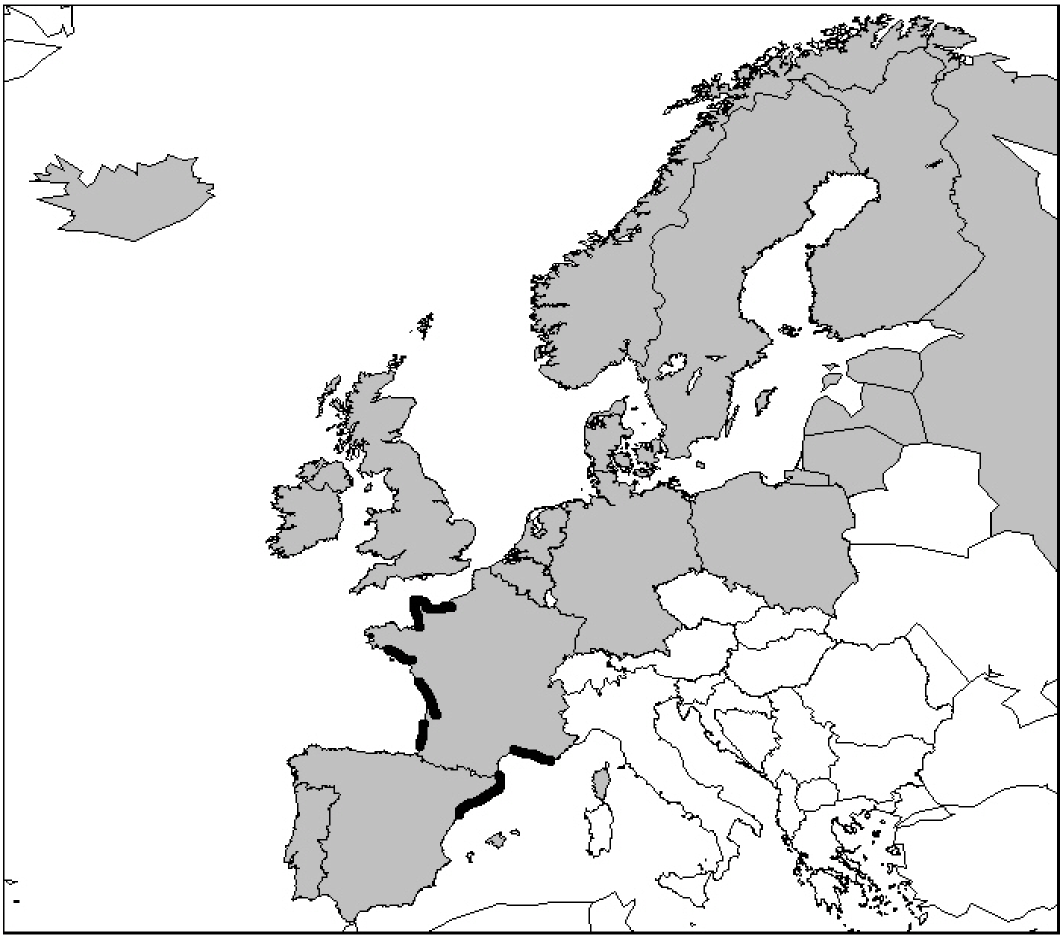

Figure A8.6. The geographical distribution of *Marteilia refringens.*

Introduction. *Marteilia refringens* is a haplosporidium affecting the digestive system of the flat oyster, *Ostrea edulis*. The parasite was observed for the first time in Normandy, France in the beginning of the 1970s and has since spread associated with transfers of oysters.

Affected oysters usually exhibit poor condition index with emaciation, discolouration of the digestive gland, cessation of growth, tissue necrosis, and mortalities. However, *Marteilia* can occur in some oysters without causing disease. The factors triggering a pathogenic host response are not clearly established, but may be related to environmental stresses or stock differences in disease resistance. Mortality appears to be related to the sporulation of the parasite, which occurs in the epithelial cells of the digestive tubules. Earlier stages occur in the epithelia of the digestive ducts and possibly the gills.

Content of the map. The map illustrates the geographical distribution of *Marteilia refringens* (heavy black line).

Source of the data. WGPDMO members.

The geographical distribution of *Perkinsus marinus*.

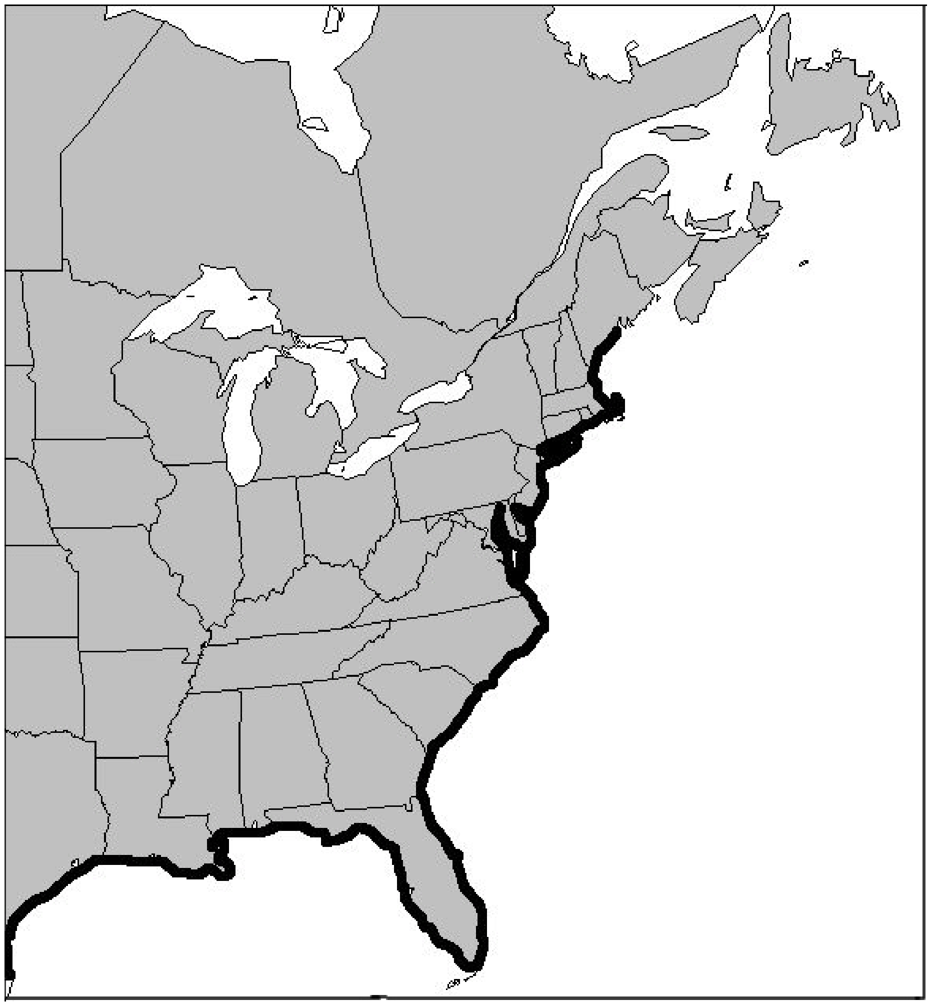

Figure A8.7. The geographical distribution of *Perkinsus marinus*.

Introduction. *Perkinsus marinus* is an intracellular parasite (2–4 µm) infecting the haemocytes of eastern oyster *Crassostrea virginica* and is one of the primary factors that adversely affects the abundance and productivity of this species.

Proliferation of the parasite causes systemic disruption of connective tissue and epithelial cells and is correlated with warm summer water temperatures (higher than 20 °C), when pathogenicity and associated mortalities are highest. Some oysters may survive summer proliferation, but are unable to revive following over-wintering dormancy. Mortalities of up to 95 % have occurred in eastern oysters during the second summer following transfer to disease enzootic areas.

Content of the map. The map illustrates the geographical distribution of *Perkinsus marinus* (heavy black line).

Source of the data. WGPDMO members.

The geographical distribution of marine VHS-like virus.

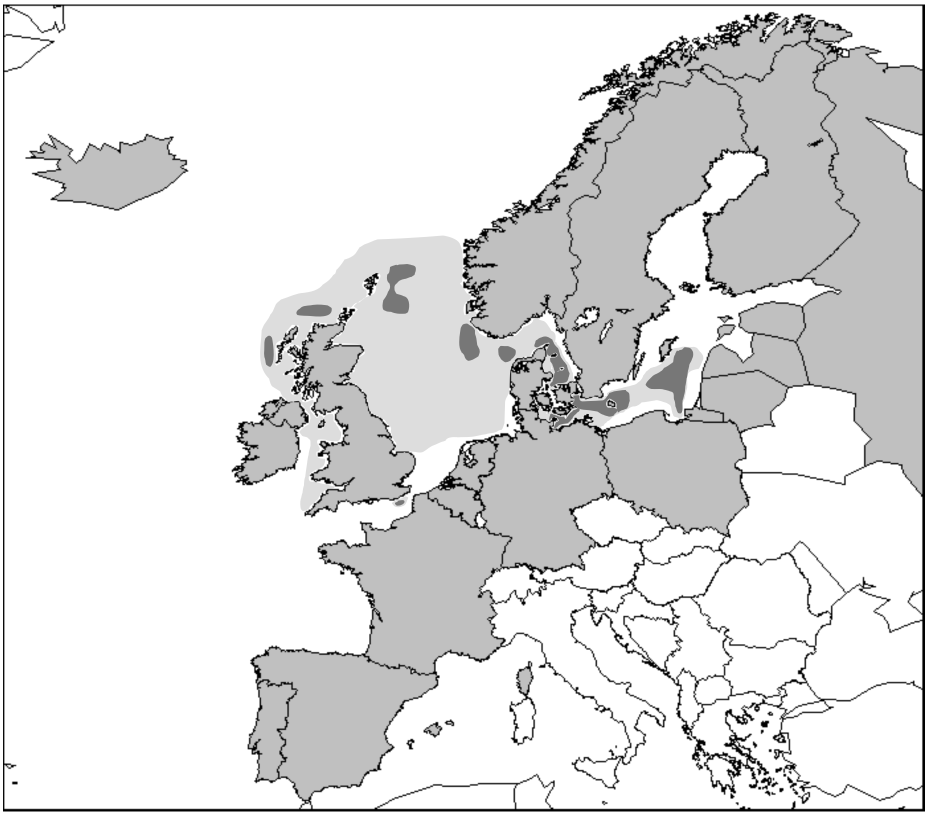

Figure A8.8. The geographical distribution of marine VHS-like virus.

Introduction. Marine Viral Haemorrhagic Septicaemia (VHS) is a disease closely related to VHS known from freshwater rainbow trout farming, in which it causes severe disease outbreaks with high mortality.

In 1988, the VHS virus was isolated for the first time from ascending Pacific salmon species. In contrast to the freshwater VHS, the marine isolates appeared to be apathogenic to rainbow trout. Further classification has revealed that the marine VHS is genetically different from the freshwater type.

The marine VHS has so far been isolated from eight different fish species in the USA and Canadian Pacific waters and in fourteen species in the North Sea and adjacent waters.

Reports from the USA suggest that marine VHS virus may be involved in epizootics in Pacific herring. The European investigations have shown relatively high prevalences of the virus in herring and sprat in the Baltic Sea. Outbreaks of marine VHS have been observed in turbot produced in aquaculture facilities in Europe and the widespread spatial distribution of the virus in the marine environment may constitute a potential threat to the aquaculture industry.

Content of the map. The map illustrates the areas investigated since 1988 for the presence of the marine VHS-like virus (light grey shading) and the areas where VHS-like virus has been isolated (dark grey).

Source of the data. Published literature and EU reports.

ANNEX 9

STRUCTURE, PROCESS, AND LIMITATIONS OF ENVIRONMENTAL ASSESSMENTS AND PRODUCTION OF ENVIRONMENTAL QUALITY STATUS REPORTS

1 INTRODUCTION

As defined by ICES (1988a),

"... [a] regional environmental assessment is a product of a rigorous and detailed review of data on conditions in the subject marine area, the objective of which is to determine the nature and severity of environmental disturbances and trends that are the consequence of anthropogenic activity."

Furthermore, the ACMP (ICES, 1988a) defined the purpose of an environmental assessment as follows:

"*The results of an environmental assessment provide the basis for strategic analysis of the requirements for regulatory action necessary to protect the marine environment in a given area, particularly for determining the adequacy and/or shortcomings of existing environmental regulations and controls pertaining to the protection of the environmental health and quality of the marine environment. It can form the basis of appropriate management plans.*"

There is now considerable experience in the Northeast Atlantic region with the conduct and international reporting of marine environmental assessments, both within OSPAR and HELCOM. None of the activities in this subject area has yet proceeded entirely smoothly, and it is possible to envisage a number of improvements which could usefully be implemented before further regional or sub-regional assessments are undertaken.

2 OBJECTIVES

It is essential that the objectives of an environmental assessment (EA) or quality status report (QSR) be clear and transparent, and agreed by all parties before the process commences. Furthermore, it should not be permissible for administrative or political considerations to allow the objectives to be modified when the data gathering and collation are already under way. We now have a better understanding of the types of objectives which are achievable given the present state of technology and knowledge, so it should be possible to develop more realistic objectives for future assessments. Examples of unrealistic objectives can easily be found among the frequent requirements to demonstrate that a particular regulatory action has led to an environmental improvement, for example, in reducing concentrations of contaminants in biota, or the recovery of populations of pelagic organisms. In both of these cases, natural variability is high, which in turn implies the need for very intensive sampling programmes. The costs of these may be very high, so the scope of the objectives should be tailored to balance the scale of the potential problem with the expected resources needed to address it. It is also futile to expect assessments to demonstrate environmental improvements if relevant regulatory actions are minimal, e.g., the impossibility of exerting serious control of introductions and transfers of organisms.

3 REPORT STRUCTURE

The structure followed in the recent OSPAR QSR 2000 and its component sub-regional assessments, as well as in earlier assessments, essentially involved a breakdown of the material by scientific discipline, i.e.,

1) Introduction and Scope;
2) Geography, Hydrography and Climate;
3) Human Activities;
4) Chemistry;
5) Biology;
6) Overall Assessment.

Although this broadly conforms to earlier ICES and GESAMP advice (ICES, 1988a, 1988b; GESAMP, 1994), in the light of experience it is now felt that this structure is too rigid, and does not permit the satisfactory treatment of a number of marine environmental issues which clearly straddle several disciplines and cannot be dealt with satisfactorily in a piecemeal fashion. The Overall Assessments which are intended to synthesize material on the various thematic issues generally fail to do justice to these matters, partly due to constraints on space.

Although it was probably appropriate to use this format for the first region-wide assessments, including such matters as detailed descriptions of geography, climate and human activities, it is considered that subsequent assessments do not need to repeat these generic subjects in their entirety. This does not, of course, preclude the need to provide regular updates on these background issues, as appropriate. Furthermore, and even more importantly, future assessments would benefit from a more thematic structure which takes each major issue and deals with all its aspects in a holistic way.

A possible revised structure of this type has been suggested by Boelens (unpublished, 2000):

6.1 Introduction
6.2 Major issues in perspective
6.3 Impacts of land-based activities
  6.3.1 Contaminant discharges
  - Hazardous substances
  - Nutrients
  - Man-made radionuclides
  6.3.2 Use of the coastal zone
  - Pressures on habitats and biodiversity
  - Tourism and recreation (including litter)
6.4 Impacts of sea-based activities
  6.4.1 Shipping (including litter)
  6.4.2 Dredging and dumping
  6.4.3 Exploitation of resources
  - Fishing
  - Mariculture
  - Oil and gas recovery
  - Aggregate extraction
6.5 Climate change
6.6 Other issues

The above is merely one suggestion and there are, of course, other ways in which a thematic structure could be arranged. However, it is felt that a holistic approach of this general type will be much more clearly understood by the public and by policy-makers, even though it may be more difficult to put together.

Another question concerns whether or not sub-regional reports are needed, or whether a single region-wide assessment is all that is required in the future. The conclusion here is that when dealing with an area as large as that covered by OSPAR and HELCOM, sub-regional reports are indeed valuable because it is easier to reflect in them the local scientific knowledge which is an essential component of a sound and comprehensive assessment. However, it is of course essential that each sub-regional report should be written to a common agreed format in order to facilitate the ultimate production of a region-wide report (see Section 5, below).

4 DATA GATHERING, ARCHIVING, PROCESSING, AND TRANSMISSION

This is perhaps one of the most intractable issues involved in the production of international environmental quality assessments because many different scientists and information handlers in a range of institutes are involved in gathering data, and they have a tendency to use a multiplicity of data storage systems and transmission protocols. Furthermore, different countries and sub-regions have given variable priorities to particular aspects of data gathering, partly due to insufficient resources and occasionally patchy expertise, and partly because some issues are not seen as being locally important or because adequate methodology is not available or was not available until recently. Especially notable is the scarcity of biological data and data on biological effects of contaminants. It goes without saying that all relevant primary monitoring data should be gathered in all relevant areas, using pre-agreed sampling and analysis protocols, and should be subject to appropriate QA/QC procedures (e.g., as agreed by QUASIMEME, BEQUALM, and within HELCOM). This should be less of a problem in the future as the OSPAR CEMP and HELCOM COMBINE programmes are now mandatory for all of their Contracting Parties.

Further problems arise, however, from the fact that input data are not always collected or are not necessarily available for the same period as the assessment. It is also particularly difficult to construct assessments of human activities in an area because the underpinning socio-economic data are not generally collected with environmental quality assessments in mind, and may be difficult or impossible to obtain, even from bodies such as EUROSTAT. Indeed, there is a fundamental distinction between the data collected as part of an agreed monitoring programme, and other data brought in by the assessors from published and other sources. The latter information is unlikely to conform to the same protocols and QA standards as those generated by the monitoring programme, so the difficulties that this will cause for assessments should be tackled at an early stage.

However, the way in which these data are subsequently handled, summarized, and stored is also a major issue for attention. Additional consideration should also be given to the more integrated collection of field data than has hitherto been the case. For example, a great deal of the biological effects data and chemical residue data used in assessments are still generally collected in an uncoordinated fashion, which in turn makes the attribution of cause-effect relationships very uncertain. Integrated monitoring of this type has been strongly advocated by ICES and OSPAR, but only a few countries have begun to implement this philosophy in actual fact.

It is considered essential that, in future assessment exercises, all people involved in data gathering and manipulation should use fully compatible spreadsheets and databases (e.g., agreed versions of Excel and Access). This applies especially to the presentation of data in map form, where the GIS software (e.g., MapInfo or ArcView) and generic base maps must be agreed in advance with both data originators and printers, and all data linked with geographical coordinates to enable their presentation via GIS systems. Finally, when e-mailing data, graphics and text, it is essential to agree the exact procedures in advance, for example, through the use of zip files in order to reduce attachments to a size acceptable for the e-mail systems used.

One further point of difficulty concerns the models used by different sub-regions for drawing conclusions about the priority issues for further action. It is clear that different models may give different views about this important matter, so it is desirable that all sub-regions harmonize their prioritization procedures.

The comments above have implications for the costs of producing assessments; it is important that the costs of arranging for harmonized software are taken into account at the start of the process.

5 PROCESS

The way in which most sub-regional or regional assessments have been produced essentially involves large numbers of scientists and other specialists in drafting specific pieces of text which then have to be welded together into seamless reports. All authors generally have to follow pre-arranged guidelines concerning the structuring of their text. This process has proved extremely difficult to manage, and in practice the guidelines are widely ignored and the people who have the task of preparing the final reports waste much time in trying to reconcile incompatible contributions from the various authors.

A much more successful model, which has been implemented in a few areas, is to give the task of preparing first drafts of environmental quality assessments to small dedicated teams, either experts working within a government institute in a "lead" country, or competent consultants working under contract. This avoids the difficulty of welding together incompatible text, and produces an initial "straw man" which is then circulated for comments to all relevant bodies. At the very least, the structure "guidelines" to be followed by the various authors should be termed compulsory instructions, and not optional extras which can be ignored or changed while the assessment process is under way. It can be helpful to restrict the scope for individual variability of authors by requiring them to provide their written contributions on a pre-determined electronic template, and their data in suitable spreadsheet templates.

While assessment drafting is in progress, it is important that all stakeholders should have ready access to the source data (scientific and socio-economic) and to the textual material being compiled, so that data gaps and misinterpretations can be rapidly detected and eliminated. One possible way forward would be for the drafting team to place all text, tables, and figures on a continuously updated website that is freely accessible to all relevant parties. Comments on early drafts would have to be routed back to the drafting team in a controlled and structured way, but it is considered that an on-line facility of the type outlined would simplify the difficult job of both drafters and reviewers. At the same time, all data used in the assessment should be placed on a central data management system (for example, at ICES) which can be accessed on-line, and which is provided with standardized tools for extracting and manipulating data.

It is clear that in previous assessment exercises, deadlines for submission of material or comments have regularly been missed. This is partly a question of discipline and individual responsibility, but mainly one of resource limitation. Many of the personnel that have in the past been involved in generating data and drafting assessments have not been allocated sufficient ring-fenced time in which to do the work. In many cases, the assessment activities have had to be fitted into an already over-full schedule, but this is clearly unacceptable. Apart from the practical considerations, this has given administrators the false impression that assessments can be produced more cheaply than is actually the case. As with the supply of compatible software, the allocation of sufficient fully-funded staff time is crucial to the successful production of quality assessments.

Some consideration should be given to the frequency with which regional and sub-regional assessments are produced. Gaps of several (e.g., five) years between reports mean that valuable expertise developed during the previous exercise will be lost, and the whole process will probably have to start from scratch. This is clearly undesirable, especially when one considers that administrators ideally need up-to-date information, not material which is several years old. It is much easier to update a report produced within the last year or two, than to start writing one *de novo*, so consideration should be given by marine monitoring organizations to the production of more frequently updated environmental quality assessments. Alternatively, specific thematic reports could be brought out annually (with different themes being treated each year) or even more frequently in certain cases, e.g., harmful algal blooms, while scientific background reports could be produced at less frequent intervals (5–10 years).

Finally, it is considered that the provision by ICES of scientific peer reviews for external quality assurance of assessments should continue. However, it should be noted that the recent peer review performed on the OSPAR QSR 2000 was a very protracted and inefficient process due to the fact that the text being considered was far from complete. It will be important in future peer reviews to ensure that draft QSRs are not submitted to ICES until they are considered complete in all respects apart from final editorial formatting. It may also be appropriate for ICES to advise on the planning of assessments, and even to contribute to some assessments, but this raises conflict-of-interest issues which would need to be carefully managed.

6 LIMITATIONS OF MARINE ENVIRONMENTAL ASSESSMENTS

It is clear that there are a number of fundamental limitations in our knowledge, understanding, and techniques which should be overcome in order to improve future environmental assessments. These are listed below, in no particular order of priority:

1) As indicated above, there is a certain amount of unevenness in the availability of monitoring expertise and techniques in various countries. This is partly

being addressed through the QUASIMEME and BEQUALM programmes, but it should be remembered that these activities are primarily concerned with QA and not with basic training.

2) There is a widespread shortfall in the availability of data on trace organic contaminant inputs to marine areas. This is partly because of the impossibility of monitoring for all organic materials, but also due to patchy availability of data held in various more or less obscure databases. More generally, most countries hold good quality monitoring data that, for various reasons, have not been submitted to ICES for use in environmental assessments. These data are often held in a scattered form and some may not be retrievable without the expenditure of resources.

3) The generation of long-term time series of physical, chemical, and biological data which can be used for predicting the impact of climate variability and human activities on the marine environment is not always being properly funded by ICES Member Countries.

4) With respect to the ecosystem effects of fisheries, there is still very poor information on multispecies interactions, impacts of demersal fishing gears, and seasonal/spatial variability in species composition of discards. Furthermore, there is uncertainty about how to decide whether fishery stocks are outside safe biological limits, and what the environmental implications may be.

5) The data available on the economic value of marine resources are very inadequate.

6) The derivation and use of Background/Reference Concentrations (B/RCs), Ecotoxicological Assessment Criteria (EACs), and Ecological Quality Objectives (EcoQOs) used by OSPAR require further refinement and thought. B/RCs and EACs currently provide a less than perfect basis for evaluating marine contaminant monitoring data because they fail to take account of bioavailability and mixtures, while outline EcoQOs now need to be operationalized through the development of more detailed and precise wording.

7) Although the concept of integrated monitoring of contaminant and biological variables has been accepted by OSPAR and other monitoring organizations, this has as yet not been fully implemented by contracting parties in their national monitoring programmes. This severely limits the interpretation of chemical monitoring data (for example, through the use of EACs), and often prevents identification of the chemical causes of biological impacts.

8) There is a general shortage of biological, and biological effects, data from outside the fisheries sector. This limits our ability to assess the scale and severity of anthropogenic and natural environmental changes.

7 RECOMMENDATIONS

1) When organizing future assessment exercises, it is recommended to HELCOM and OSPAR that responsibility for the initial drafting of sub-regional and regional reports should be given to small, properly funded teams working to a clearly defined and mandatory format. Furthermore, draft text, tables, and figures should be placed on websites where they can be continuously reviewed by all relevant contributors. Finally, all data, maps, and text submitted to, or generated by, drafting teams must be organized in a standardized and pre-arranged format.

2) It is recommended that all relevant monitoring guidelines (covering sampling, analysis, and reporting) be agreed and finalized before monitoring begins. Such guidelines should be mandatory.

3) It is recommended as a matter of urgency that ICES update its data management system so that it can provide an effective on-line facility for storing all data used in future marine environmental assessments. Such a facility must include on-line tools for the extraction and manipulation of data in a standard format. Furthermore, all relevant contributors of data must have easy access to all the information held in this way.

4) When modern database facilities go on line, all ICES Member Countries should be encouraged to submit the many monitoring data sets which are held nationally but have not yet been made available for marine environmental assessments.

5) ICES Member Countries should ensure that the training, protocols, and equipment required to implement international monitoring programmes are universally available.

6) In collaboration with appropriate national and international agencies, ICES Member Countries and monitoring Commissions should start planning immediately for future regional assessments, with special emphasis on the development of arrangements for obtaining data that are not being generated within their monitoring programmes (e.g., information on the economic value of marine resources). In this connection, assessment procedures may have to be designed to fulfill the requirements of the Commissions, EU regulations, and perhaps also individual Member Countries.

7) ICES Member Countries should be encouraged to continue funding the generation of long-term data sets which can be used for predicting the impacts of climate change on the marine environment. Future marine environmental quality assessments will be

seriously incomplete if such data sets are not continuously expanded.

8) It is recommended that OSPAR, HELCOM, and ICES give further thought to the development and use of B/RCs and EACs. In particular, it should be recognized that these tools cannot on their own provide reliable guidance on the environmental risks posed by contaminant concentrations. The linked monitoring of chemical and biological measures of contaminants and their effects should therefore be a central principle of future assessments. Although the concept of integrated monitoring has been accepted by the monitoring organizations, there is insufficient awareness among contracting parties that this implies the need to coordinate sampling programmes to the extent that chemical and biological measurements are made on the same samples wherever possible and appropriate.

8 DOCUMENTS CONSIDERED

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9 REFERENCES

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ANNEX 10

PREPARATION OF ICES REPORTING FORMATS FOR BIOLOGICAL EFFECTS DATA

This annex contains a list of measurements to be included in the development of reporting formats for data on biological effects of contaminants. Table A10.1 lists the data reporting requirements that are similar for all biomarkers. In Table A10.2, protein measurements have been kept separate from the other biomarkers, although they would obviously need to be linked to the appropriate biomarker measurement. Tables A10.3 to A10.7 concern data measurements specific to individual biomarkers. No table for imposex/intersex measurements is provided here as further work is required to complete the list of data requirements for these methods.

It should be noted that it is vital that information on individual fish can be linked so that all information concerning one individual or pooled sample, including histopathology, contaminant concentrations, fish morphometry, and biomarker responses can be retrieved. Bioassay results must be linkable to data on the relevant matrix.

Table A10.1. General data requirements for all biomarkers.

Super Group	Sub-Group	Specifics	ICES code	Format number
Sampling	Logistics	Site identification	multiple	01 CF
		Date of sampling	SDATE	01 CF
		Sample size	NOINS	01 CF
		Fish species	SPECI	01 CF
		Number of samples collected		
		Salinity		
		Description of abnormalities	ASTSA?/ ~DISEA	01? CF/ 08 DF
		Bottom temperature at time of capture		
	Subsample	Sex	SEXCO	04 CF
		Size	LNXXX	04 CF
		Age		
		Length	LNXXX	04 CF
		Weight	WTXXX	04 CF
		Liver weight	TISWT	07 CF
		Method for estimating liver weight		
		Gonad weight	GONWT	04 CF
		Maturation status – juvenile, maturing, ripe		
		Gross external disease or parasites	ASTSA?/ DISEA	01? CF/08 DF
		Compliance to international quality control programme (BEQUALM, QUASIMEME)		
		Intralaboratory quality assurance (accreditation, GLP)		

Table A10.2. Data requirements specific to protein measurements.

Super Group	Sub-Group	Specifics	ICES code	Format number
Method and assay	Tissue	Tissue used	TISSU	07 CF
		Preparation analysed – microsomes/cytosol/S9/whole tissue		
	Type of assay	Define assay*		
		Protein standard		

*Alternatives for "define assay" are: Lowry, Bradford, Smith, and other.

Table A10.3. Data requirements specific to EROD.

Super Group	Sub-Group	Specifics	ICES code	Format number
Method and assay	Tissue	Tissue used	TISSU	07 CF
		Length of storage – tissue		
		Temperature of storage	METSP	20 CS
	Conditions	Length of storage – prepared sample		
		Temperature of storage – prepared sample		
		Preparation analysed – microsomes/S9 fraction		
		Define assay*	~METAN	21 CW
		Detection method – fluorometric, spectrophotometric		
		Instrumentation – single-cell, plate-reader		
		Substrate solvent type		
		Substrate solvent concentration		
		Use of protease inhibitor		
		Detection limit, as implemented in practice	DETLI	21 CF
		Assay temperature	INCTE	22 CF
		Ethoxyresorufin concentration	SUBCO?	22? CF
		NADPH concentration in assay	Rename COFCO?	22 CF
		pH in assay	INCPH	22 CF
		Internal or external standard used		
		Concentration range for calibration curve of external standard		
		Concentration of internal standard spike		
		Buffer concentration in assay		
		Protein concentration in assay	PROTC?	22? CF
		Type of buffer	NABUF	22 CF
		Resorufin purity (molar absorbance)		
Results		EROD activity (pmol/min/mg protein)		

*Allowed entries for "define assay": kinetic, endpoint.

Table A10.4. Data requirements specific to Oyster Embryo Bioassay.

Super Group	Sub-Group	Specifics	ICES code	Format number
Sampling	Logistics	Depth of sediment		
		Origin of oysters	OOYST	23 CS
		Month of collection	MOYST	23 CS
		Conditioning of oysters	COYST	23 CS
		Spawning natural/artificial		
		Number of replicates		
Method and assay		Depth	WADEP	01 CS
		Sediment quality (grain size, organic content)		
		REDOX in sample		
		Reference toxicant		
		Salinity		
		Number of samples		

Table A10.5. Data requirements specific to metallothionein.

Super Group	Sub-Group	Specifics	ICES code	Format number
Method and assay	Tissue	Tissue used	TISSU	07 CF
		Length of storage time – tissue		
		Temperature of storage	METSP	
	Conditions	Length of storage time – prepared sample		
		Temperature of storage		
		Define method*		
		Reducing agent in homogenization		
		DPP – temperature and duration heat treatment		
		Temperature in assay		
		Denaturing method (where relevant)		
		Nature of buffer	NABUF	22 CF
		Standard**		
		ELISA/RIA – primary antibody type and source		
		Saturation assays – radioactive or non-radioactive (Hg, Cd, Ag assays)		
		DPP – electrolyte composition (deviation from standard)		
		Chromatographic – elution buffer		
		Chromatographic – protein assay		
		Detection limit, as implemented in practice	DETLI	21 CF
		Use of protease inhibitor		

*Allowed entries for “define method”—DPP, ELISA, RIA, Hg-saturation, Cd-saturation, Cd/Chelex, Ag-saturation, spectrophotometric, chromatographic, other.
**Allowed entries for “standard”—MT (from same species), mammalian MT, GSH, other.

Table A10.6. Data requirements specific to DNA adducts.

Super group	**Sub-group**	**Specifics**	**ICES code**	**Format number**
Sampling		Fish identifier		
	Sub sample	Tissue	TISSU	07 CF
Storage		Length of storage time		
		Temperature of storage time		
Method	^{32}P-postlabelling	Detection limit, as used in practice	DETLI	21CF
		DNA purification		
		DNA digestion		
		DNA determination		
		DNA enrichment procedure		
		Extraction of adducts		
		32P source		
		32P specific activity in assay		
		Labelling procedure		
		Type of TLC sheet		
		Separation procedure		
		Quantification method		
		RNAse treatment		
		alpha-amylase treatment		
		Protease treatment		
		Check ATP following postlabelling		
		Control labelling step		
		Ratio of DNA adducts detected to amount DNA used (nmol DNA adducts/mol DNA)		
		Level of DNA adducts in farmed salmon testes blank		
		Level of DNA adducts in positive control		

Table A10.7. Data requirements specific to whole sediment bioassays, using *Corophium volutator* or *Arenicola marina*.

Super group	**Sub-group**	**Specifics**	**ICES code**	**Format number**
Type of test		Bioassay or toxicity test		
Storage prior to assay		Length of storage time	METSP	20CS
		Temperature of storage time	METSP?	20CS
Bioassay		Date of assay		
		Sediment particle size	SFRAC?	10CS
		Ammonia in sediment		
		Redox in sediment at collection		
		pH in sediment at collection		
		Salinity in sediment at collection		
		Depth of sediment		
		Number of replicates (sample, control)	NUMCR/ NUMSR	23CS
		Number of animals used		
		Origin of specimens		
		Size of animals		
		Reference toxicant		
		Sample manipulations		
		Volume of replicate		

ANNEX 11

ACME/ACMP ADVICE BY TOPIC FOR THE YEARS 1989–2000

Numbers in the table refer to sections of the present report and of the ACMP or ACME reports from 1989 to 2000, in reverse chronological order.
*Signifies major advice on that topic.

Topic	Sub-topic	2000	1999	1998	1997	1996	1995	1994	1993	1992	1991	1990	1989
Monitoring	Strategy						5.1	*4; *Ann. 1	5	5.1			
	Programme evaluation	6.7						4.2					
	Statistical methods for design	6.6.2	5.6										
	Benthos	6.8.1				6.1.2; 11.1; *Ann. 8				8; *Ann. 6	8.1	9	*Ann. 1
	NSTF/MMP								5.2			9.3	4
	Sediments/guidelines			4.6; *Ann. 2	4.5; *Ann. 1	5.5; *Ann. 4		5.5	6.1; *Ann. 1				*14
	Sediment data normalization	6.5.1; Ann. 1	5.5		4.5.2	5.5.1		5.5					*14.1
	Sediment sensitivity, variance factors						5.6						
	Metals/sediments					9.5	5.6	5.5					
	Matrix tables												
	▪ general (JMP)											*Ann. 1	*6.1
	▪ organic											6.1	
	▪ NSTF											6.1	
	Substances that can be monitored												
	▪ organic		5.4	4.5		5.4	6.6	6.8					
	▪ inorganic	6.4	5.4	4.5	4.2								
	Use of seaweeds							5.1				6.8	6.3
	Use of seabird eggs		13.2; Ann. 7	4.7.5									
	Spatial monitoring	6.6.3			*4.7.2		5.3	5.1					
	JAMP/JMP guidelines				4.1	5.2;5.4	5.4			13.3			
	BMP guidelines					5.1.2	5.4		5.3				
	AMAP	6.3	5.2	4.4		5.1.3		5.4					
	Effects of nutrient enrichment		12.1			9.1	5.8						
	Monitoring PAHs			4.2; *Ann. 1	4.4.1; 4.5; *Ann. 1								

Topic	Sub-topic	2000	1999	1998	1997	1996	1995	1994	1993	1992	1991	1990	1989
Assessment	Statistical methods			4.7.1									
	Combined effects of contaminants		5.1.5	11.2									
Temporal trend monitoring	Strategy/objectives					*4; Ann. 1		*4; *Ann. 1					
	Guidelines					4.4		4;5.2					
	Data analysis			4.7.4	15.2			5.2	6.2	5.2			
	Nutrients					5.7			6.3				
	Fish/JMP/JAMP		5.6.4			5.6					5.1	6.2	
	Fish/CMP					5.6					5.1	6.5	*6.4
	Biota/BMP						7.3						
	Biological effects			4.1									
	Mussels										5.1	*Ann. 3	
	Pooling											6.7; Ann. 5	6.4.3
	Precision	6.6.5						5.2		*Ann. 1			6.4.4
	Sediment storage												
	Sediments					4.3; 5.5.3							
	Statistical requirements			4.7			4.3						
Integration of biological/ chemical measurements	Sediment quality					5.2.2	4.2; Ann. 2	5.4	6.4; *Ann. 2				
Biological effects monitoring	Monitoring strategy					*5.3	4.1; *Ann. 1						
	Concepts												*7.1
	Statistical design			4.1*	4.31								
	Methods	6.1.2	5.1.1		4.3.2	5.3.2	Ann. 1			6.2	7.2		
	Molecular techniques						*5.2						
	Pathology				4.3.3	5.3.3	8.4	9.4					
	Workshop results	6.1.4				5.3.2				6.1	7.1	8.1	7.2
	Source of variability	6.1.1; 6.6.4	5.1.2										
	Data analysis												
	▪ general									6.3			
	▪ EROD									*Ann. 2			
	▪ oyster bioassay									*Ann. 2			
	Endocrine disruption	9.6											

Topic	Sub-topic	2000	1999	1998	1997	1996	1995	1994	1993	1992	1991	1990	1989
Baseline studies	ICES Baseline TM/SW										6	*7	6.5
	Contaminants in ▪ Baltic sediments		5.2	4.3	6.1	7.1	7.1	7.1	8.1	13.2	14.1	15.1	
	▪ North Sea sediments									13.1			
	HCH in sea water									14			
Regional assessments	Guidelines											5	
	Preparation plans	18.4; Ann. 9											5
	North Sea QSR								4.1	4	4	5	
	Baltic Sea					7.2	7.2	7.3					5
	Baltic fish	18.2				7.3	7.2	7.3				17.2	17.3
	Canadian waters										16		
	Nutrient trends–North Atlantic											13	12
Quality assurance	Philosophy										13.6		
	Reference materials	7.5; Ann. 4		5.6	4.2			*6.9	7.11				13.1
	Oxygen in sea water				*Ann. 3							14.5	13.6
	Nutrients				5.7								
	Quality/comparability ▪ organic contaminants		5.4	4.5			*6.6	*6.8					
	Hydrocarbons												13.7
	Lipids						6.4	6.5					
	NSTF											14.7	
	Biological effects techniques	7.3	7.4	5.4	5.3	*6.2; *Ann. 5	6.2		7.1		7.3		
	Sediment quality criteria											15.2	22.2
	QA of sampling	7.7	7.6	5.7	5.10					*12.8			
	QA info. in data bank				16.1.1			6.10					
	Chemical measurements–Baltic Sea	7.4	7.5	5.5	5.4	6.3	6.3	6.2	7.4				
	Biological measurements	7.1; 7.2	7.1; 7.2	5.1; 5.2	5.1; 5.2	6.1	6.1	6.1	7.3				
	Fish disease monitoring			8.2	5.3.2	*Ann. 6							

Topic	Sub-topic	2000	1999	1998	1997	1996	1995	1994	1993	1992	1991	1990	1989
Intercomparison exercises	Status		Ann. 8		Ann. 10	Ann. 10	Ann. 7	Ann. 6	Ann. 5	Ann. 8	Ann. 3	Ann. 9	Ann. 2
	Nutrients/sea water					6.4	6.5	6.6	7.8	12.4		14.1	13.4
	Hydrocarbons in ▪ biota						6.7						13.7
	▪ sediments												13.7
	▪ sea water												13.7
	PAHs/standards									12.2	13.1	14.2	13.2
	PCBs/CBs in biota						6.6; 6.7	6.3;6.4	7.5	12.1	13.2		
	Organochlorines in biota						6.6; 6.7						
	CBs/standards											14.3	13.3
	CBs in sediments						6.6	6.3	7.5		13.2		
	Metals in ▪ sea water				5.5	6.5							
	▪ SPM							6.7	7.9	12.3	13.3	14.4	13.5
	Dissolved oxygen in sea water									12.5	13.4	*14.5	
	Oyster embryo bioassay									Ann. 4			
	EROD									12.6; *Ann. 3			
Methods	Nutrients in sea water												13.5
	DO in sea water				*Ann. 3								
	Sediment normalization	6.5.1; Ann. 1						5.5			14.2; 14.3		
	Organic carbon measurements				4.6; Ann. 2								
	Zooplankton studies	6.8.3											
Algal blooms	Primary production methods								6.5	11	11.1	12.1	
	Initiating factors										*11.3	12.2	
	Dynamics			10.2	9.2			8	10				
	Exceptional blooms	6.8.2; Ann. 3	12.2; Ann. 2	Ann. 3	Ann. 8						11.2		
	Phycotoxins/ measurements										11.4	12.3	
	C. polylepis bloom												*11.1

Topic	Sub-topic	2000	1999	1998	1997	1996	1995	1994	1993	1992	1991	1990	1989
Fish diseases and related issues	Relation to pollution			8.2	8.3	5.3.3	8.4	9.4			9.1		9.3
	Survey methods				7.2						9.2		
	Diseases in wild fish	9.1	10.1										
	Baltic fish	9.4; *Ann. 6			7.2		8.1; 8.2; 8.3	9.3					
	Survey results												9.1
	Data analysis	9.2	10.2; Ann. 5	8.1; *Ann. 8	7.1	8.2		9.5	9.4	7			
	M74 in Baltic salmon	9.3	10.3	8.3	6.2	7.4	7.4	9.1					
Mariculture	Interactions	14.1	15.2			15.1	14	13		9.1		*11	10
	Escape of fish—effects	15.1		14.2	14.1								
	Nutrient inputs/Baltic	14.2								*9.2		11.1	10.4
	Use of chemicals	14.1.2	15.2		14.2							Ann. 6	
Introductions and Transfers	Code of Practice					14.2	13.1	14.1	12.1				
	Accidental transfers, including via ships	10.1.2; 10.2	11.1	9.1; 9.2	13.2	14.4; *Ann. 9	13	14.2	12.3				
	Genetically modified organisms		15.1	9.3		14.5			12.2				
	On-going introductions	10.1.1			13.1	14.1							
	Baltic Sea		11.1	9.3		14.3							
Marine mammals	Contaminants/effects	13.2	14.1	12.2; *Ann. 10	11.4	5.4.2; 13.3; 13.4							
	Seal epidemic 1988											*18	*18.1
	Baltic marine mammal stocks	*13.1	14.2		11.1	13.1		10.2		*18			
	Populations/N. Atlantic							10.1	11.1	*18	18		
	Pathogens								11.2; Ann. 3				
	Impact of fisheries			12.1; *Ann. 9	11.2; 11.3	13.2							
Seabirds	Diet, food consumption		13.1; *Ann. 6										
	Use in contaminant monitoring		13.2; *Ann. 7				5.3						
	Effects of contaminants	12.1											
Overviews	Mercury			7.1; *Ann. 4									*19.1
	Hormone disruptors			7.4; Ann. 6		Ann. 2							
	HCB											*20.1	
	Lindane (γ-HCH)											*20.1	

Topic	Sub-topic	2000	1999	1998	1997	1996	1995	1994	1993	1992	1991	1990	1989
Overviews (cont.)	Tris(4-chlorophenyl) methanol/methane	8.1				10; *Ann. 7							
	Benzene/ alkylated benzenes						10.2; *Ann. 5						
	Chlorinated alkanes	8.1					10.1; Ann. 4						
	PBBs and PBDEs		9.2; *Ann. 4										
	PCDDs and PCDFs												*19.2
	PCDEs				8.1; *Ann. 6								
	TBT		9.2; *Ann. 3										
	Octachlorostyrene										20.1		
	Toxaphene				8.1; *Ann. 5			12.3					
	Atrazine							12.1					
	Irgarol 1051				8.1; *Ann. 4								
Classification/ assessment tools	Human health											19	
	Hazardous substances			11.1			12.3			*15			
	Background concentrations	6.2			15.1		12.1						
	Ecological Quality Objectives—North Sea	18.6	17.2										
	Ecotoxicolgical reference values						12.2						
	Environmental indicators		17.1										
Sand/gravel extraction	Code of Practice											16	
	Effects	16	8.1	6.1	*6.3						15	16	15
	Environmental impact assessment		8.1				*15	*15	13				
Modelling	Radioactive contaminants/Baltic Sea									*17.1			
	Use in monitoring and assessment							16		17.2			

Topic	Sub-topic	2000	1999	1998	1997	1996	1995	1994	1993	1992	1991	1990	1989
Data banks and management	Nutrients	20.2	19.2	15.2	17.2	16.1.2							
	Contaminants	20.1	19.1	15.1	17.1	16.1.1; 16.3	17	2.2	2.2				
	NSTF									20	*21	22	
	ICES format	20.1.10; 20.5	19.1			16.6							
	ICES databases	20.1	19.1						14				
	Biological database	20.3	19.3	15.1.3	17.3; 17.4			11.2; Ann. 4					
	AMAP	20.1.3	19.1.3		17.1.1	16.2							
Ecosystem effects of fishing	General	18.1			*12	12		18		*19	19		
	Effects of disturbance on benthos	5		10.4	9.3	11.2	9; Ann. 3	11.1		8.3	8.2		
	Seabird/fish interactions	12.2	4		10			19					
	Changes in abundance of non-target fish species			*13.3									
	Models and metrics	18.5		13.4.1									
	Effects on level of predation on benthos by fish			13.4.2									
	Impact on size/age and spatial distributions of target fish			13.1									
	Discards	*11		13.2									
Inputs of contaminants and nutrients	Riverine inputs ▪ gross			4.7.2; 4.7.3	*4.7.1								
	▪ net												16
	Trend detection methods	*6.6.1; *Ann. 2	*6; *Ann. 1										
	Atmospheric inputs				4.7.1								
ICES Environmental Report	Oceanographic conditions	18.3.1	8.4.1	6.2.1									
	Zooplankton			6.2.2									
	Harmful algal blooms	18.3.2	8.4.2; Ann. 2	6.2.3; Ann. 3									
	Fish disease prevalence	18.3.3; Ann. 8											

Topic	Sub-topic	2000	1999	1998	1997	1996	1995	1994	1993	1992	1991	1990	1989
Special topics	Context of ACMP advice											Ann. 7	*21
	Patchiness in Baltic Sea												17.1
	Nutrient trends/eutrophication in OSPAR area				Ann. 9							13	12
	Nutrients and eutrophication			10.1	9.1	9.1	5.8		6.3	10	*11.3		
	Sediments—Baltic		5.2	4.3	6.1	7.1	7.1	7.1			14.1	15.1	14.2
	Sediments ▪ bioavailability					9.3	4.2; Ann. 2	5.4	6.4; Ann. 2		7.4		7.3
	▪ release of contaminants												14.3
	Bioaccumulation of contaminants				8.2; *Ann. 7								
	Acid rain studies/effects												20
	Coastal zone fluxes								8.2				
	Influence of biological factors on contaminant concentrations			7.2; Ann. 5									
	Discharge of produced water by offshore platforms			7.6; Ann. 7									
	North Sea Benthos Survey						9			*Ann. 5			
	GLOBEC	19.1	18.1	16.2									
	GOOS	19.2	18.2	16.1	16								
	Marine habitat classification/ mapping	17	16										
	Toxicity of dredged material	8.4											

ANNEX 12

TITLES OF RECENTLY PUBLISHED ICES COOPERATIVE RESEARCH REPORTS

No.	Title
217	Report of the ICES Advisory Committee on the Marine Environment, 1996
218	Atlas of North Sea Benthic Infauna
219	Database Report of the Stomach Sampling Project, 1991
220	Guide to Identification of North Sea Fish Using Premaxillae and Vertebrae
221	Report of the ICES Advisory Committee on Fishery Management, 1996 (Part 1 and Part 2)
222	Report of the ICES Advisory Committee on the Marine Environment, 1997
223	Report of the ICES Advisory Committee on Fishery Management, 1997 (Part 1 and Part 2)
224	Ballast Water: Ecological and Fisheries Implications
225	North Atlantic-Norwegian Sea Exchanges: The ICES NANSEN Project
226	Report on the Results of the ICES/IOC/OSPARCOM Intercomparison Programme on the Determination of Chlorobiphenyl Congeners in Marine Media—Steps 3a, 3b, 4 and Assessment
227	Tenth ICES Dialogue Meeting
228	Report of the 11th ICES Dialogue Meeting on the Relationship between Scientific Advice and Fisheries Management
229	Report of the ICES Advisory Committee on Fishery Management, 1998 (Part 1 and Part 2)
230	Working Group on Methods of Fish Stock Assessment—Reports of Meetings in 1993 and 1995
231	Status of Introductions of Non-Indigenous Marine Species to North Atlantic Waters 1981–1991
232	Diets of Seabirds and Consequences of Changes in Food Supply
233	Report of the ICES Advisory Committee on the Marine Environment, 1998
234	Report of the Workshop on Ocean Climate of the NW Atlantic during the 1960s and 1970s and Consequences for Gadoid Populations
235	Methodology for Target Strength Measurements (with special reference to *in situ* techniques for fish and micronekton)
236	Report of the ICES Advisory Committee on Fishery Management, 1999 (Part 1 and Part 2)
237	Seventh Intercomparison Exercise on Trace Metals in Sea Water
238	Report on Echo Trace Classification
239	Report of the ICES Advisory Committee on the Marine Environment, 1999
240	Report of the Young Scientists Conference on Marine Ecosystem Perspectives

ACRONYMS

ACFM	Advisory Committee on Fishery Management
ACG	Assessment Coordination Group (OSPAR)
AChE	acetylcholinesterase
ACIA	Arctic Climate Impact Assessment
ACME	Advisory Committee on the Marine Environment
ACMP	Advisory Committee on Marine Pollution
ADI	acceptable daily intake
AHH	aryl hydrocarbon hydroxylase
AI	artificial intelligence
ALA-D	δ-aminolevulinic acid dehydratase
AMAP	Arctic Monitoring and Assessment Programme
ANOVA	analysis of variance
AQC	analytical quality control
ARC	Aquatic Restoration and Conservation
ASC	Annual Science Conference (ICES)
ASC	Assessment Steering Committee (AMAP)
ASCOBANS	Agreement on Small Cetaceans of the Baltic and North Sea
ASE	accelerated solvent extraction
ASG	Assessment Steering Group (AMAP)
ASMO	Environmental Assessment and Monitoring Committee (OSPAR)
ASP	amnesic shellfish poisoning
ATHN	7-acetyl-1,1,3,4,4,6-hexamethyltetrahydro-naphthalene
ATP	adenosine triphosphate
AUV	automated underwater vehicle
BACI	before/after comparison impact
BCF	bioconcentration factor
BCPS	bis-*p*-chlorophenyl sulfone
BCR	European Commission Community Bureau of References
BEQUALM	Biological Effects Quality Assurance in Monitoring Programmes
BEWG	Benthos Ecology Working Group
BFG	Institute of Hydrology (Koblenz, Germany)
BMB	Baltic Marine Biologists
BMP	Baltic Monitoring Programme (HELCOM)
BP	before present
B/RC	background/reference concentrations
BSH	Federal Maritime and Hydrographic Agency (Germany)
BSPAs	Baltic Sea Protected Areas
CAFF	Conservation of Arctic Flora and Fauna
CBs	chlorobiphenyls
CCA	canonical correspondence analysis
CDEs	chlorodiphenylethers
CD-ROM	compact disc: read-only memory
CEFAS	Centre for Environment, Fisheries and Aquaculture Science (UK)
CEMP	Coordinated Environmental Monitoring Programme (OSPAR)
CFP	ciguatera fish poisoning
CHBs	chlorinated bornanes
CHCs	chlorinated hydrocarbons
CIEM	Conseil International pour l'Exploration de la Mer
CIL	Cambridge Isotope Laboratories (USA)
CMDGC	comprehensive multidimensional gas chromatography
CITES	Convention on International Trade in Endangered Species
CMP	Cooperative ICES Monitoring Studies Programme
COMBINE	Cooperative Monitoring in the Baltic Marine Environment (HELCOM)
CORINE	EEA Coordination of Information on the Environment
CPR	Continuous Plankton Recorder
CPUE	catch per unit effort
CRIMP	Centre for Research on Introduced Marine Pests (Australia)
CRMs	certified reference materials
CTD	conductivity–temperature–density
CUSUM	**Cu**mulative **Sum**
CV	coefficient of variation
DBT	dibutyltin

DCA	Danish Coastal Authority
DCM	dichloromethane
DDE	dichlorodiphenylethylene
DDT	dichlorodiphenyltrichloroethane
∑DDT	total DDT
DES	diethylstilbestrol
DETR	Department of the Environment Transport and Regions (UK)
DG	Directorate General
DIFFCHEM	OSPAR Working Group on Diffuse Sources
DMA	dimethylarsenic
DNA	deoxyribonucleic acid
DO	dissolved oxygen
DOC	dissolved organic carbon
DPP	differential pulse polarography
DPSIR	driving forces-pressure-state-impact-responses
DR-CALUX	dioxin-responsive chemical-activated luciferase
DSP	diarrhetic shellfish poisoning
DST	diarrhetic shellfish toxin
DTA	direct toxicity assessment
DYNAMEC	Ad Hoc Working Group on the Development of a Dynamic Selection and Prioritisation Mechanism for Hazardous Substances (OSPAR)
EAC	ecotoxicological assessment criteria
EC	European Commission
EC MON	Environment Committee Working Group on Monitoring and Assessment (HELCOM)
ECD	electron capture detection
ECE LRTAP	Economic Commission for Europe Long-Range Transboundary Air Pollution Convention (UN)
EcoQO	ecological quality objective
ED	endocrine disruptor
EDCs	endocrine-disrupting chemicals
EEA	European Environment Agency
EI	electron impact ionization
EIA	environmental impact assessment
ELISA	enzyme-linked immunosorbent assays
EMD	evaporative mass detector
ENDS	Environmental Data Services (UK)
ENSO	El Niño Southern Oscillation
EOC	elemental organic carbon
EOCl	extractable organic chlorine
EPA	Environmental Protection Agency (USA)
EQG	environmental quality guidelines
EQS	environmental quality standard
ER-CALUX	oestrogen-responsive chemical-activated luciferase
EROD	ethoxyresorufin-*O*-deethylase
ESB	Environmental Specimen Bank (Germany)
ESE	enhanced solvent extraction
ETC/MCE	European Topic Centre on Marine and Coastal Environment
EU	European Union
EUNIS	European Nature Information System
EUROSTAT	Statistical Office of the European Communities
EUT	Ad Hoc Working Group on Eutrophication (OSPAR)
FAO	Food and Agriculture Organization
FDA	fluresein diacetate
FDE	Fish Disease Data Entry Program
FID	flame ionization detection
FMO	flavin-containing mono-oxygenase
FRS	Fisheries Research Service (UK)
FTZ	Research and Technology Centre of Kiel University (Germany)
GABA	γ-aminobutyric acid
GAM	general additive model
GC	gas chromatography
GC/ECD	gas chromatography/electron capture detection
GC/MS	gas chromatography/mass spectrometry
GESAMP	Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection
GEOHAB	Global Ecology and Oceanography of Harmful Algal Blooms (IOC-SCOR)
GIS	Geographical Information System
GLM	general linear model
GLOBEC	Global Ocean Ecosystem Dynamics Programme
GOOS	Global Ocean Observing System

GPC	gel permeation chromatography
GPX	glutathione peroxidase
GSI	gonadosomatic index
GST	glutathion-*S*-transferase(s)
HAB	harmful algal bloom
HAE	harmful algal event
HAEDAT	Harmful Algal Event Database
HARP	Harmonized Reporting Procedures (OSPAR)
HELCOM	Helsinki Commission (Baltic Marine Environment Protection Commission)
HPLC	high performance liquid chromatography
HRMS	high resolution mass spectrometry
HSP	heat shock protein(s)
IASC	International Arctic Science Committee
IBTS	International Bottom Trawl Survey
ICES	International Council for the Exploration of the Sea
IFREMER	Institut Français de Recherche pour l'Exploitation de la Mer
IGBP	International Geosphere-Biosphere Programme
IJC	International Joint Commission (Canada and the USA)
IMM	Intermediate Ministerial Meeting (North Sea)
IMO	International Maritime Organization
IMPACT	Working Group on Impacts on the Marine Environment (OSPAR)
IMPACT II	EU-funded project on the impact of bottom trawls on the North Sea and Irish Sea benthic ecosystems
INPUT	Working Group on Inputs to the Marine Environment (OSPAR)
IOC	Intergovernmental Oceanographic Commission
IOS	Initial Observing System (GOOS)
IOW	Institute of Baltic Research (Warnemünde, Germany)
IPCC	Intergovernmental Panel on Climate Change
ISA	Infectious Salmon Anaemia
ISE	Intersex Sequence Index
ISO	International Organization for Standardization
ITIS	Interagency Taxonomic Information System (USA)
IUCN	International Union for the Conservation of Nature and Natural Resources
IWC	International Whaling Commission
JAMP	OSPAR Joint Assessment and Monitoring Programme
JGOFS	Joint Global Ocean Flux Study (IGBP)
JMP	OSPAR Joint Monitoring Programme
JNCC	Joint Nature Conservation Committee (UK)
LD	lethal dose
LGC	Laboratory of the Government Chemist (UK)
LIFE	EC conservation programme
LMR	Living Marine Resources Panel (GOOS)
LOESS	statistical smoother
LOWESS	statistical smoother
LOI	loss-on-ignition
LPS	Laboratory Performance Scheme (QUASIMEME)
LRMs	laboratory reference materials
LRMS	low resolution mass spectrometry
MA	Marketing Authorisation
MAFF	Ministry of Agriculture, Fisheries and Food (UK)
MAHs	monocyclic aromatic hydrocarbons
MarLIN	Marine Life Information Network (UK)
MAST	Marine Science and Technology Programme (EC)
MBT	monobutyltin
MCWG	Marine Chemistry Working Group
MDA	malone dialdehyde
MDGC	multidimensional gas chromatography
MDR	multidrug resistance
MEDPOL	Monitoring and Research Programme of the Mediterranean Action Plan
MFO	mixed-function oxidase
MMA	monomethylarsenic
MMHg	monomethylmercury
MMP	Monitoring Master Plan (NSTF)
MON	Ad Hoc Working Group on Monitoring (OSPAR)

MPA	Marine Protected Area
mRNA	messenger ribonucleic acid
MS	mass spectrometry
MSD	mass selective detector
MSY	maximum sustainable yield
MT	metallothionein
mtDNA	mitochrondrial DNA
MXR	multixenobiotic resistance
NAO	North Atlantic Oscillation
NCI	negative chemical ionization
NCM	Nordic Council of Ministers
NEUT	Working Group on Nutrients and Eutrophication (OSPAR)
NIES	National Institute for Environmental Standards (Japan)
NIHS	National Institute of Health Science (Japan)
NIST	U.S. National Institute of Standards and Testing
NIVA	Norsk instittutt for vannforskning [Norwegian Institute for Water Research]
NMBAQC	National Marine Biological Analytical Quality Control Scheme (UK)
NMR	nuclear magnetic resonance
NOAA	National Oceanic and Atmospheric Administration (USA)
NOAEC	no observable adverse effects concentration
NOAEL	no observable adverse effects level
NODC	National Oceanographic Data Center (USA)
NOEL	no observable effects level
NORWECOM	**Norw**egian **Eco**logical **M**odel
NOWESP	North-West European Shelf Programme (EU MAST Project)
NRC	National Research Council (Canada)
NRR	neutral red retention
NSP	neurotoxic shellfish poisoning
NSTF	North Sea Task Force
NWRI	National Water Research Institute (Canada)
OCs	organochlorines
OCPs	organochlorine pesticides
OECD	Organisation for Economic Cooperation and Development
OIE	Office International des Epizooites
OM	oxidizable matter
OPs	organophosphates
OSPAR	OSPAR Commission
PAHs	polycyclic aromatic hydrocarbons
PAMP	Post-Authorisation Monitoring Programme (Scotland)
PAR	photosynthetic available radiation
PBBs	polybrominated biphenyls
PBDEs	polybrominated diphenylethers
PBTs	persistent, bioaccumulative, toxic compounds
PCA	principal component analysis
PCBs	polychlorinated biphenyls
PCDDs	polychlorinated dibenzo-*p*-dioxins
PCDEs	polychlorinated diphenylethers
PCDFs	polychlorinated dibenzofurans
PCNA	proliferating cell nuclear antigen
PCNs	polychlorinated naphthalenes
PCP	pentachlorophenol
PCR	polymerized chain reaction
PCTs	polychlorinated terphenyls
PEEK	polyetheretherketone
PFC	plaque-forming cell
PHAHs	polyhalogenated aromatic hydrocarbons
PICES	North Pacific Marine Science Organization
PICT	pollution-induced community tolerance
PNEC	predicted no-effect concentration
POC	particulate organic carbon
POPs	persistent organic pollutants
PROD	pentoxyresorufin-*O*-deethylase
PSP	paralytic shellfish poisoning
PSU	practical salinity unit
PTFE	polytetrafluorethene
QA	quality assurance
QC	quality control
QSARs	quantitative structure–activity relationships
QSR	quality status report

QUASIMEME	Quality Assurance of Information for Marine Environmental Monitoring in Europe
QUASH	Quality Assurance of Sampling and Sample Handling (EC)
RIA	radioimmunoassay
RIKZ	Rijksinstituut voor Kust en Zee [National Institute for Coastal and Marine Management]
RMs	reference materials
RPSI	Relative Penis Size Index
ROSCOP	Cruise Summary Report
ROV	remotely operated vehicle
RQSR	regional Quality Status Report
RUBIN	Rutin för Biologiska Inventeringar
RV	research vessel
SACs	special areas of conservation
SAR	species at risk
SCOR	Scientific Committee on Oceanic Research
SEPA	Scottish Environment Protection Agency
SFE	supercritical fluid extraction
SFG	scope for growth
SG	Study Group
SGBOSV	ICES/IOC/IMO Study Group on Ballast and Other Ship Vectors
SGDIB	Study Group on Estimation of the Annual Amount of Discards and Fish Offal in the Baltic Sea
SGEAM	Study Group on Ecosystem Assessment and Monitoring
SGMHM	Study Group on Marine Habitat Mapping
SGPHYT	Study Group on an ICES/IOC Checklist of Phytoplankton
SGQAB	ICES/HELCOM Steering Group on Quality Assurance of Biological Measurements in the Baltic Sea
SGQAC	ICES/HELCOM Steering Group on Quality Assurance of Chemical Measurements in the Baltic Sea
SGQAE	ICES/OSPAR Steering Group on Quality Assurance of Biological Measurements related to Eutrophication Effects
SIM	selected ion monitoring
SIME	Working Group on Concentrations, Trends and Effects of Substances in the Marine Environment (OSPAR)
SMLIPA	ICES Special Meeting on the Use of Liver Pathology in Flatfish for Monitoring Biological Effects of Contaminants
SOAEFD	Scottish Office Agriculture, Environment and Fisheries Department
SOD	superoxide dismutase
SOPs	Standard Operating Procedures
SP	solid phase
SPM	suspended particulate material
SPME	solid-phase micro-extraction
SRMs	standard reference materials
TALs	total annual loads
TBA	tetrabutylammonium
TBPS	tertiary butylbicyclophosphorothionate
TBT	tributyltin
TCDD	tetrachlorodibenzo-*p*-dioxin
TCPM	*tris*(4-chlorophenyl)methanol
TCPMe	*tris*(4-chlorophenyl)methane
ΣTCP	TCPM + TCPMe
TDC	Thematic Data Centre
TE	toxic equivalent
TEF	toxic equivalency factor
TEQ	toxic equivalent
TIE	toxicity identification evaluation
TIMES	*ICES Techniques in Marine Environmental Sciences*
TIP	training and intercalibration programme
TMA	trimethylarsenic
TMAP	Trilateral Monitoring and Assessment Programme
TOC	total organic carbon
TON	total oxidized nitrogen
TPT	triphenyltin
UK	United Kingdom
UN	United Nations
UNEP	United Nations Environment Programme
UNCED	World Commission on Environment and Development

UNESCO	United Nations Educational, Scientific, and Cultural Organization
U.S.	United States
USA	United States of America
USEPA	United States Environmental Protection Agency
UV	ultraviolet
VDSE	Vas Deferens Sequence Index
VEN	viral erythrocytic necrosis
VHS	viral hæmorrhagic septicæmia
VIC	Voluntary International Contaminant Monitoring in Temporal Trends (OSPAR)
VMD	Veterinary Medicines Directorate (UK)
VOCs	volatile organic compunds
VPC	Veterinary Products Committee (UK)
VTG	vitellogenin
WCRP	World Climate Research Programme
WGAGFM	Working Group on the Application of Genetics in Fisheries and Mariculture
WGBAST	Baltic Salmon and Trout Assessment Working Group
WGBEC	Working Group on Biological Effects of Contaminants
WGECO	Working Group on Ecosystem Effects of Fishing Activities
WGEIM	Working Group on Environmental Interactions of Mariculture
WGEXT	Working Group on the Effects of Extraction of Marine Sediments on the Marine Ecosystem
WGHABD	ICES/IOC Working Group on Harmful Algal Bloom Dynamics
WGITMO	Working Group on Introductions and Transfers of Marine Organisms
WGMDM	Working Group on Marine Data Management
WGMMHA	Working Group on Marine Mammal Habitats
WGMMPD	Working Group on Marine Mammal Population Dynamics and Trophic Interactions
WGMS	Working Group on Marine Sediments in Relation to Pollution
WGPDMO	Working Group on Pathology and Diseases of Marine Organisms
WGPE	Working Group on Phytoplankton Ecology
WGSAEM	Working Group on the Statistical Aspects of Environmental Monitoring
WGSE	Working Group on Seabird Ecology
WGSSO	Working Group on Shelf Seas Oceanography
WGZE	Working Group on Zooplankton Ecology
WKQAC	Second ICES/HELCOM Workshop on Quality Assurance of Chemical Procedures for the COMBINE and PLC-4 Programmes
WOCE	World Ocean Circulation Experiment
WWW	world wide web