ICES COOPERATIVE RESEARCH REPORT

RAPPORT DES RECHERCHES COLLECTIVES

NO. 263

Report of the ICES Advisory Committee on the Marine Environment, 2003

Copenhagen, 16–20 June 2003

International Council for the Exploration of the Sea

Conseil International pour l`Exploration de la Mer

Palægade 2–4 DK-1261 Copenhagen K Denmark

December 2003

For purposes of citation, the 2003 ACME Report should be cited as follows:

ICES. 2003. Report of the ICES Advisory Committee on the Marine Environment, 2003. ICES Cooperative Research Report, 263. 227 pp.

ISBN 87-7482-008-7
ISSN 1017-6195

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ICES ADVISORY COMMITTEE ON THE MARINE ENVIRONMENT

LIST OF MEMBERS

16–20 June 2003

Participant	Affiliation
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J. Rice	Chair, Consultative Committee
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[1] Unable to attend.

Secretariat members participating in the meeting or portions thereof:

D. Griffith	ICES General Secretary
M. Sørensen	Data Manager

EXECUTIVE SUMMARY

The ICES Advisory Committee on the Marine Environment (ACME) met from 16 to 20 June 2003. As part of its work during this period, the ACME prepared responses to the requests made to ICES by the OSPAR Commission and 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 relevance for ICES Member Countries as well as of interest to the Commissions and other readers of this report.

As a result of the creation of the Advisory Committee on Ecosystems (ACE), several topics previously handled by ACME have been moved to the remit of ACE and scientific information and advice on these topics can be found in the ACE report for 2003. The topics covered include ecosystem effects of fishing, ecological quality objectives, ecosystem modelling and assessment, marine mammals issues, biodiversity issues, and marine habitat classification and mapping.

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

Introductions and Transfers of Marine Organisms

An important outcome of the meeting was the adoption of a revised Code of Practice on the Introductions and Transfers of Marine Organisms, to update the 1994 Code of Practice (Section 9.1 and Annex 7). ICES recommends that Member Countries adopt and apply this revised Code of Practice for use in mariculture and other activities involving the introduction or transfer of non-native species, and commends it to the OSPAR and Helsinki Commissions for their use also.

The ACME reviewed information on the imports of live aquatic species in ICES Member Countries for aquaculture, restocking, and live food sales (Section 9.2). The most commonly moved species in 2002 continued to be Atlantic salmon (*Salmo salar*) and Pacific oysters (*Crassostrea gigas*). The proposed introduction of the Asian oyster *Crassostrea ariakensis* into tidal waters of the Chesapeake Bay was also reviewed.

Brief reviews of the impact of three accidentally introduced species are provided (Section 9.3). These cover the introduction of the club tunicate (*Stylea clava*) on the Atlantic coast of Canada, the introduction of the slipper limpet (*Crepidula fornicata*), and the ectoparasite (*Gyrodactylus salaris*) in Sweden.

Issues relevant to the transfer of organisms via ships' ballast water and hulls are reviewed in Section 9.4. This material concludes that currently implemented ballast water control and management measures are restricted to ballast water exchange at sea, and there is an urgent need to find alternative, more effective, and environmentally sound ballast water treatment methods.

Monitoring

In 2003, the ACME continued work on the development of biological effects monitoring programmes. The ACME reviewed progress in the analysis of results from the Sea-going Workshop on Pelagic Biological Effects Methods (BECPELAG) that was conducted during 2001 (Section 4.1.1 and Annex 1). It also revised tables of recommended and promising methods for biological effects monitoring (Section 4.1.2) and reviewed the use of Toxicity Identification and Evaluation (TIE) procedures in monitoring programmes (Section 4.1.3). Noting the need to prepare guidelines for integrated chemical and biological effects monitoring, the ACME summarized the background for this issue and recommended a way forward (Section 4.2).

The ACME also considered progress in the development of procedures for monitoring temporal trends of contaminants in sediments (Section 4.3) and accepted an inventory of national programmes for trend monitoring of contaminants in sediments (Annex 2), which is provided for OSPAR and HELCOM for use in their trend assessments of contaminants in sediments.

In relation to the EU Water Framework Directive (WFD), the ACME prepared advice on the preferred matrix for monitoring priority compounds under the WFD (Section 4.4). Finally, the ACME reviewed ICES activities in relation to the Global Ocean Observing System (GOOS) (Section 4.5).

Quality Assurance and Intercomparison Exercises

In relation to the quality assurance of biological measurements in the Baltic Sea, the ACME reviewed the results of the work on this topic during the past year and provided advice for the Helsinki Commission (Section 5.1). This advice includes the recommendation that the institutes and countries around the Baltic Sea continue active implementation and harmonization of QA procedures for biological monitoring and that they actively develop software to permit the transfer of national data to the ICES Biological Data Reporting Format for submission to the ICES Marine Data Centre.

For the OSPAR area, the ACME reviewed progress in the implementation of quality assurance procedures for biological measurements (Section 5.2). Noting that OSPAR had ceased financial support for this work during 2003, the ACME recommended that such support should again be provided by OSPAR. In Section 5.3, the ACME took note of the outcome of the EU-funded project Biological Effects Quality Assurance in Marine Monitoring (BEQUALM) and its future plans as a self-funding programme.

With regard to chemical measurements, five technical notes were accepted as 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 5.4). These technical notes cover quality assurance procedures for the determination of heavy metals, chlorobiphenyls, and polycyclic aromatic hydrocarbons (PAHs) in sediments and for PAHs in biota, as well as a technical note on the validation of an established analytical method for chemistry.

Methodologies and Results of Environmental Assessments

A review of approaches for setting Sediment Quality Guidelines and an inventory of Sediment Quality Criteria values and standards for dredged material disposal in ICES Member Countries is provided in Section 6.1 and Annex 3; the latter is recommended for consideration by OSPAR and HELCOM for use in their assessments of sediment quality and dredged material. A method to test whether an annual mean contaminant concentration is at or below a reference value is presented in Section 6.2.1 and Annex 4; this can be used for near-background concentrations as well as for the classifications for ecological status employed in the EU Water Framework Directive.

The ACME also reviewed a proposal for a standard assessment protocol for trends in contaminant concentrations that had been presented at the 2002 meeting of the OSPAR Working Group on Monitoring. In particular, the effect of outliers on trend detection and the effect on trend detection of allocating different values for undetected observations were explored (Section 6.2.2). Based on this, advice was provided for the next OSPAR temporal trend assessment of data. Advice to supplement that provided in previous years' ACME reports was presented concerning the analysis of monthly and quarterly data on inputs of nutrients and contaminants to the marine environment. This particularly covers the situation when there is a break in the slope of the trend line in the time series (Section 6.3.1 and Annex 5) for the extraction of anthropogenic signals from environmental data influenced by meteorological conditions (Section 6.3.3).

Section 6.4 provides comments on statistical problems experienced in relation to the development of environmental indicators, while Section 6.5 provides some statistical considerations regarding the development of methodology for joint assessments of input data and data on concentrations of contaminants in biota and sediments. Finally, summaries of the ICES Environmental Status Report for oceanographic conditions, plankton monitoring results, harmful algal blooms, and fish disease prevalence are provided in Section 6.6.

Contaminants in the Marine Environment

The ACME reviewed the OSPAR List of Chemicals for Priority Action and provided advice on the state of the analytical methodology for analysing each chemical, or group of chemicals, in marine environmental samples (Section 7.1) as well as on the availability of methods to monitor the biological effects of these substances at environmental concentrations. New information is provided on the following contaminants: 1) polybrominated diphenylethers (PBDEs) (Section 7.2.1); and 2) phenylurea herbicides (diuron and isoproturon) (Section 7.2.2 and Annex 6).

Report sections responding to requests specific to the OSPAR Commission

OSPAR List of Chemicals for Priority Action

The ACME reviewed the OSPAR List of Chemicals for Priority Action and provided advice on the state of the analytical methodology for analysing each chemical, or group of chemicals, in marine environmental samples (Section 7.1) as well as on the availability of methods to monitor the biological effects of these substances at environmental concentrations. An extensive list of references for analytical methodologies for these substances has also been included.

Eutrophication Status of the Marine Area

Advice in relation to data products for developing the OSPAR Common Procedure for Identification of the Eutrophication Status of the Maritime Area is presented in Section 14.3. The advice from 2002 is repeated, together with an urgent request to countries to submit their nutrients data to support the next OSPAR thematic assessment for eutrophication in 2006..

Implementation of the OSPAR Joint Assessment and Monitoring Programme

At the request of the OSPAR Commission, the ACME reviewed the implementation table for the OSPAR Joint Assessment and Monitoring Programme and indicated the types of scientific activities required to fulfill each item on the table and the actions that ICES could carry out to assist in this implementation (Section 13).

Effects of Extraction of Marine Sand and Gravel on Marine Ecosystems

The ACME reviewed marine extraction activities in ICES Member Countries during 2002 as well as approaches to environmental impact assessment and related environmental research, as summarized in Section 12.2. A brief review is provided on the application of risk assessment methods as a tool for the management of marine extraction activities (Section 12.3).

The ACME also reviewed and accepted revised ICES Guidelines for the Management of Marine Sediment Extraction (Section 12.1 and Annex 10), which take into account comments from OSPAR on the previous version of these Guidelines that had been adopted in 2002.

Data Handling

The annual review of data handling activities by the ICES Marine Data Centre on contaminants data relevant to the requirements of OSPAR, HELCOM, and AMAP is contained in Section 14.1 of this report. This section also includes a brief overview of the structure of the new environmental database in ICES. Section 14.2 summarizes the work of the ICES Marine Data Centre in handling nutrients data relevant to the OSPAR programmes.

Report sections responding to requests specific to the Helsinki Commission

Data products on nutrients in the Baltic Sea

In addition to the website prepared by ICES in 2002 that provides an inventory of data on nutrients in the Baltic Sea, including tables with the numbers of measurements of each parameter at Baltic Monitoring Programme stations, and a facility to plot temporal trends of mean concentrations, ICES has prepared indicator reports on temporal trends in contaminant concentrations in herring and on chlorophyll *a* in sea water. This is reviewed in Section 14.1.3.

Advice and information on topics of general interest

Fish Diseases

An overview of new trends in the occurrence of diseases in wild and farmed fish and shellfish stocks is contained in Section 8.1. Viral Haemorrhagic Septicaemia virus continues to increase its host spectrum, distribution, and occurrence, particularly on the west coast of North America. Two new mycobacterial species have been identified from striped bass in Chesapeake Bay, USA, with one of these species found in 76% of striped bass. *Gyrodactylus salaris* remains a major threat to Atlantic salmon in Norway, while the sea louse *Lepeophtheirus salmonis* has been implicated in unusually low returns of pink salmon in western Canada and continues to be a major problem for wild salmon and sea trout in Norway.

The M74 syndrome in Baltic salmon continues to increase in most Baltic rivers compared to the preceding two years (Section 8.2). A review of strategies to assess the prevalence of shellfish diseases in parallel with fish diseases and chemical contaminant levels is provided in Section 8.3. Several case studies are provided and it is noted that the combined use of fish and shellfish in monitoring studies is advantageous, given the different vulnerability of fish and shellfish species to various groups of contaminants.

Section 8.4 contains a review of the impact of diseases of farmed fish on wild fish stocks. It is pointed out that the transfer of disease may occur from farmed fish to wild fish but also from wild fish to farmed fish. Examples are given of several major diseases and parasites that may be transferred in either direction. Specific studies are recommended to assess particular aspects of this problem. The effectiveness of management control methods for sea lice in salmon farms in ICES Member Countries is summarized in Section 8.5.

Marine Biological Communities, Processes, and Responses

A review of progress in the North Sea Benthos Project and the results of statistical analyses of benthos community data are contained in Section 10.1 (plus Annex 8 for the latter). Progress in understanding the dynamics of harmful algal blooms is reported in Section 10.2.

Environmental Assessments

Contributions to the ICES Environmental Status Report for 2002 have been made concerning oceanographic conditions (Section 6.6.1 and http://www.ices.dk/status/clim0001), plankton (Section 6.6.2), harmful algal blooms (Section 6.6.3 and http://www.ices.dk/status/decadal/), and fish and shellfish disease prevalence (Section 6.6.4 and http://www.ices.dk/status/fish_and_shellfish_diseases/index.htm).

Issues Related to Mariculture

With regard to the potential environmental interactions of mariculture, the ACME prepared information and advice concerning the potential impact of escaped non-salmonid candidates for aquaculture on local native stocks (Section 11.2). This included three examples: sea bass (*Dicentrarchus labrax*), seabream (*Sparus aurata*), and cod (*Gadus morhua*).

A review of the implications of the Water Framework Directive in EU Member States on the sustainability of aquaculture in coastal and transitional waters is provided in Section 11.1 and Annex 9. ICES encourages Member Countries with mariculture operations in coastal and transitional waters to pay particular attention to these implications.

Global Programmes

The ACME reviewed recent activities by ICES for the North Atlantic in relation to the Global Ocean Observing System (GOOS), particularly the plans for an ICES-EuroGOOS North Sea Ecosystem Pilot Project (NORSEPP) (Section 4.5). This pilot project will initially concentrate on physical oceanography in relation to fish stocks.

Sources of Information Considered by the ACME at its 2003 Meeting

At its 2003 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
PGNSP	ICES-EuroGOOS Planning Group on the North Sea Pilot Project
SGBOSV*	ICES/IOC/IMO Study Group on Ballast and Other Ship Vectors
SGGOOS	ICES/IOC Steering Group on GOOS
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*	Steering Group on Quality Assurance of Biological Measurements related to Eutrophication Effects
WGBEC	Working Group on Biological Effects of Contaminants
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
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
WGZE	Working Group on Zooplankton Ecology

*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 status and outlook for the marine environment, including contaminants, as well as a range of other environmental issues, as may be requested by ICES Member Countries, other bodies within ICES, relevant regulatory Commissions, and other organizations.

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 or advice 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 in 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, the Council created a new Advisory Committee on Ecosystems (ACE), with the primary responsibility to provide scientific information and advice on the status and outlook for marine ecosystems, and on exploitation of living marine resources in an ecosystem context. Accordingly, some of the issues that ACME has previously considered have been transferred to ACE. Thus, the ACME report will no longer contain sections on issues regarding seabirds or marine mammals, unless the material pertains to contaminants and their effects, nor on marine habitat classification and mapping or ecosystem assessment.

A summary of the progress on the 2003 programme of work requested by the OSPAR Commission is given below, along with reference to the relevant sections and annexes of this report (or of the 2003 report of the ICES Advisory Committee on Ecosystems, where applicable), where more detailed information and advice 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 brief summary of the ICES advice below in normal print.

A SCIENTIFIC ADVICE

1 Analytical methods and environmental data on chemicals identified for priority action

Provide advice on whether there are suitable analytical methods available to allow the measurement of environmental concentrations or effects of the substances listed on the OSPAR list of chemicals for priority action (as amended by OSPAR 2002), and whether any information exists on the presence of these chemicals in the marine environment.

Advice on the status of suitable analytical methods for the determination of environmental concentrations of substances on the OSPAR List of Chemicals for Priority Action, as well as on whether there are suitable methods to monitor their effects, is provided in Section 7.1 of this report. For biological effects, only methods that can be applied to field-collected organisms have been included, and only where an environmentally relevant dose may cause effects. Particular attention is given to endocrine disruption as a biological effect.

2 Assistance in the next JAMP assessment of temporal trends and levels of contaminants in sediment and biota

Provide advice on how to develop a practical scheme for carrying out assessments, using software developed by ICES, of the OSPAR monitoring data on the ICES environmental database. This scheme is required so that an assessment of the OSPAR data currently held on the ICES database can be undertaken by late 2004.

1. *Develop software for retrieving and summarising the OSPAR data from the ICES database into appropriate annual indices. This retrieval should include facilities to automate the assessment process as far as possible, e.g., use QA information directly in filtering data.*
2. *Develop software for applying trend assessments for this summarised data.*
3. *Prepare data set of relevant QA-screened data provided by ICES. This data set will be distributed for further additional expert analysis to investigate the initial analysis and its conclusions in depth;*
   a. *make a proposal on the format in which screened data will be provided.*
4. *Prepare a set of initial runs, i.e., a working document that will be ready for further analysis by experts. This working document would be based on running the developed software on all of the data sets for each contaminant for all of the OSPAR sub-regions. The initial runs should produce output for each contaminant in the form of·*
   a. *a trend assessment of each time series (i.e., of the data for a specific site);*
   b. *multiple-plots-to-a-page of all of the time series (together with fitted trends and reference lines) for all sites in an OSPAR region;*
   c. *a meta analysis (maps) of the numbers of significant trends, comparison of last year(s) to a "reference" concentration like EAC or BRC, etc., for all of the time series for all sites in an OSPAR region. (The details of the meta analyses will be refined by MON).*
5. *Prepare a final round of runs producing tables and graphs for the report. These tables and graphs should be produced with the same software. ICES to agree with MIG, based on the analysis by the scientists, a standard set of tables and figures.*

This work is being conducted by the ICES Marine Data Centre and is not reflected in this report. This work is conducted in collaboration with the OSPAR Working Group on Monitoring (MON) Intersessional Group (MIG) and will culminate with the temporal trend assessment of data on contaminants in biota and sediments in the OSPAR area in late 2004.

3 Further work to develop Ecological Quality Objectives

This activity relates to the issues, Ecological Quality Elements, and Ecological Quality Objectives agreed in the Ministerial Declaration of the Fifth International Conference on the Protection of the North Sea (the Bergen Declaration).

3.1 *For the EcoQOs relating to (i) spawning stock biomass of North Sea commercial fish species, (ii) seal population trends in the North Sea, (iii) by-catch in the North Sea of harbour porpoises and (iv) proportion of oiled common guillemots among those found dead or dying on beaches:*

a. develop draft guidelines (taking into account MON 01/9/1, Annex 6), including monitoring protocols and assessment methods, for evaluating the status of, and compliance with, those EcoQOs;

b. for EcoQOs (i) and (iii), propose a list of species to be covered by these EcoQOs; and

c. for all four EcoQOs, provide reference points (for commercial fish stocks) or current levels (for other species or populations), on an appropriate geographical basis, to be used as baselines against which progress can be measured;

d. reconstruct the historic trajectory of these metrics and determine their historic performance (hit, miss or false alarm) relative to the objective being measured, as a basis for deciding their relationship to management; and

e. provide advice on what management measures could be taken to help meet the EcoQOs.

3.2 Commence development, on the basis of the criteria for sound EcoQOs established by ICES in 2001, of related metrics, objectives and reference levels for the EcoQOs relating to (v) utilisation of North Sea breeding sites of seals, (vi) mercury concentrations in eggs and feathers of North Sea seabirds, (vii) organochlorine concentrations in seabird eggs, (viii) plastic particles in the stomachs of North Sea seabirds, (ix) local availability in the North Sea of sandeels for black-legged kittiwakes, (x) seabird population trends in the North Sea as an index of seabird community health, (xi) changes in the proportion of large fish and hence the average weight and average maximum length of the fish community, (xii) density of sensitive (e.g., fragile) species, (xiii) density of opportunistic species, and (xiv) presence and extent of threatened and declining species in the North Sea:

a. for EcoQ element (xi), develop draft guidelines (taking into account MON 01/9/1, Annex 6), including monitoring protocols and assessment methods, for evaluating the status of, and compliance with, those EcoQOs;

b. for EcoQ elements (xii) and (xiii), identify possible species in the respective categories, consider further the spatial scale requirements of sampling and the adequacy of existing monitoring activities to determine their status and trends, and provide further advice based on scenario considerations on the applications of possible EcoQOs;

c. for EcoQ element (xiv), consider the invertebrate and fish species and the habitats on the Draft OSPAR list of threatened and declining species for their relevance and usefulness as a basis for EcoQOs for the North Sea; and

d. where possible and appropriate for EcoQ elements (xi), (xii), (xiii) and (xiv), reconstruct the historic trajectory of the metrics and determine their historic performance (hit, miss or false alarm) relative to the objective being measured, as a basis for deciding their relationship to management.

This request was handled by the Advisory Committee on Ecosystems (ACE), and the response is contained in Section 6 of the 2003 ACE report. ACE reviewed general aspects of the Ecological Quality–Ecological Quality Objective (EcoQ–EcoQO) framework and prepared a template for the review of the Pilot Project on EcoQOs in the North Sea by ICES Working Groups in 2004. ACE then considered the individual EcoQ elements in the OSPAR request, and prepared a detailed review of, and advice on, the four EcoQ elements listed in item 3.1 of the above request. Initial consideration and advice is also provided for the nine EcoQ elements listed in item 3.2 of the request.

The four EcoQ elements in item 3.1 of the request were subjected, to the extent possible, to a performance analysis using the approach of signal-detection and decision theory, as a means of determining their value in a management context. Based on these analyses, advice is provided on the further development of these EcoQOs.

A review of the EcoQ elements and EcoQOs listed in item 3.2 shows that some are in an advanced stage of development, while others may prove difficult to implement in a useful way. Advice is given in the 2003 ACE Report for the continued development of these EcoQs.

B DATA HANDLING

4 Carry out data handling activities for CEMP data

This activity covers the following data types:

a) contaminant concentrations in biota and sediments;

b) measurements of biological effects;

c) the implementation of the Nutrient Monitoring Programme;

d) data on phytobenthos, zoobenthos and phytoplankton species.

The ICES Marine Data Centre has handled the data submitted in 2002 covering monitoring activities in 2001. However, to date only a few biological community data have been submitted for OSPAR purposes. For data on nutrients, there continue to be problems with the timely submission of data. Further information is contained in Section 14 of this report.

5 Continue with the development of a relational database for data on contaminants in biota, sediments and water

This activity should cover the following issues:

- *Conversion to relational database;*
- *Conversion of screening program;*
- *User-friendly data and information import/export facilities;*
- *Development of programs for data products, including web inventories.*

This activity will be undertaken from 2001 through 2003, to prepare a relational database for the contaminants data, including user-friendly input and output facilities. To best meet OSPAR requirements, e-mail groups and electronic bulletin boards will be established with appropriate representation from ASMO working groups to ensure the reporting of all required meta-data as well as to initially define output products.

Considerable work on the development of the relational database for data on contaminants in marine media and biological effects of contaminants has been conducted during the past year. The incorporation of data on biological effects of contaminants has proved to be particularly challenging owing to the diverse nature of the various biological effects measurements and their reporting requirements. Detailed consideration has also been given to the integration of oceanographic (bottle water) data and the environmental data in a single database. Furthermore, the large amount of information required on methods and quality assurance complicates the data structure. In general, there seems to be a need for developing a generally agreed level of what should be stored in the database with regard to metadata.

Additional request

ACE has provided a final response to the additional request received from OSPAR in January 2002 for ICES to provide an assessment of the data on which the justification of the OSPAR Priority List of Threatened and Declining Species and Habitats will be based. A partial response was provided in the 2002 ACE report, with the supplemental advice in 2003 covering several non-commercial species of fish, seahorses, turtles, and marine mammals (see Section 5 and Annex 1 of the 2003 ACE report).

ACME has provided an additional response to the second extra request, which was received in April 2002, asking ICES to review and comment on the revised Joint Assessment and Monitoring Programme (JAMP) and advise where ICES work might contribute to the preparation of specific JAMP products, develop synergy between ICES and OSPAR programmes, and identify gaps in the context of the provision of an overall assessment of the quality of the marine environment. ACME conducted a review of the JAMP in 2002, but did not provide detailed comments on potential ICES contributions to the implementation of the JAMP. This was done in 2003, and ACME has provided detailed comments on the JAMP Implementation Table, including an assessment of the major scientific support required for each JAMP activity and the potential ICES contributions. This is contained in Section 13 of this report.

3 PROGRESS ON TASKS FOR THE HELSINKI COMMISSION

The present status of work on the 2003 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 (or of the 2003 report of the ICES Advisory Committee on Ecosystems, where applicable) where more detailed information and advice 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.*

Progress in the development of quality assurance procedures for biological measurements in the Baltic Sea is summarized in Section 5.1 of this report. In particular, the ICES/HELCOM Steering Group on Quality Assurance of Biological Measurements in the Baltic Sea (SGQAB) reviewed QA issues in relation to phytoplankton monitoring. A checklist on phytoplankton species in the Baltic Sea has been completed, and this checklist has been included in the ICES biological community database to support the submission of HELCOM phytoplankton data. Further developments, however, are needed in relation to the phytobenthos programme.

Five Technical Notes have been completed by the ICES/HELCOM Steering Group on Quality Assurance of Chemical Measurements in the Baltic Sea (SGQAC) for inclusion in the COMBINE Manual (Section 5.4). These are as follows: 1) Technical Note on the Determination of Heavy Metals in Marine Sediments; 2) Technical Note on the Determination of Polycyclic Aromatic Hydrocarbons (PAHs) in Sediments; 3) Technical Note on the Determination of Polycyclic Aromatic Hydrocarbons in Biota; 4) Technical Note on the Determination of Chlorobiphenyls in Sediments; and 5) Technical Note on the Validation of an Established Analytical Method for Chemistry.

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), being aware that the next report will be reported 2004, and that the evaluation is based on annual submission of data to ICES from ICES member states and other data submitted to ICES.*

This request was handled by the Advisory Committee on Ecosystems and the advice is contained in Section 2 of the 2003 ACE report. In its triennial review of the status of marine mammal populations in the Baltic marine area, ACE has pointed out that the populations and area of distribution of all species of marine mammals in the Baltic Sea are considerably smaller than they were a century ago, with harbour porpoise and grey and ringed seal populations a fraction of their pristine levels. Human activities, including fishing, hunting of seals, and the discharge of contaminants, have all impacted these populations. Currently, there is inadequate data on the by-catch of all three species of seals in the Baltic Sea, but by-catch is the most important cause of death of grey seals. The health status of grey seals is not normal, with colonic ulcers an important cause of mortality. ICES advises that the present population of grey seals is not in a favourable conservation status and the extra mortality caused by humans should be reduced. For harbour porpoises, ICES supports the recommendations of the ASCOBANS recovery plan for the Baltic Sea.

SPECIAL ISSUES

3) *As ICES has all the HELCOM hydrographic and hydrochemical data, HELCOM MONAS 3/2001 requests ICES in cooperation with Denmark to develop indicators on:*

- *Surface (0–10 m) concentrations of* $NO_3 + NO_2$ *and* PO_4 *in winter (means of January–February, January–March in the Gulf of Finland, December–March in the Gulf of Bothnia) from representative stations (selected by the Contracting Parties within the 6 basins, also coastal and/or hot spot stations), and present the results as time series plots;*
- *Surface (0–10 m) in summer (July–August) mean chlorophyll concentrations from the same representative stations.*

The response to this request has been prepared by the ICES Secretariat and the ICES Marine Data Centre, together with scientists from Denmark, and submitted directly to the HELCOM Secretariat. An overview of this response is contained in Section 14.1.3.

4) *HELCOM MONAS 3/2001 requests ICES in cooperation with Denmark and Sweden to develop relevant Baltic Sea bottom oxygen indicator(s).*

The response to this request has been prepared by the ICES Secretariat and the ICES Marine Data Centre, together with scientists from Denmark and Sweden, and submitted directly to the HELCOM Secretariat. An overview of this response is contained in Section 14.1.3.

5) *The HABITAT Group has requested ICES to include the Baltic Sea in its work on a marine habitat classification and mapping system generally accepted and covering the whole Baltic Sea area.*

ACE reviewed progress in the development of marine habitat classification systems, notably the EUNIS classification, and marine habitat mapping (see Section 7 of the 2003 ACE report). ICES recommends the continued development of classification systems, including continued development of the EUNIS system. ICES endorsed data management initiatives to archive metadata held by relevant agencies, as they are critical to the development of habitat maps and associated information. With respect to the Baltic Sea, this is a dynamic environment and the development of a classification scheme for this area will be best achieved through a dedicated project. Work is ongoing in cooperation with the European Environment Agency to develop such a project and obtain funding for its implementation.

6) *In connection with the standing request HELCOM HABITAT additionally requests a monitoring programme for estimation of the abundance of seal and other mammal populations.*

ACE has provided advice on specific survey methods to estimate populations of each of the three species of seals in the Baltic Sea (see Section 3 of the 2003 ACE report). Given the very low abundance of harbour porpoises in this area, a step-wise approach is recommended to obtain information on their abundance.

4 MONITORING TECHNIQUES AND GUIDELINES

4.1 Biological Effects Techniques

4.1.1 Results of the ICES Sea-Going Workshop on Biological Effects of Contaminants in Pelagic Ecosystems (BECPELAG)

Request

This is part of continuing ICES work to review progress regarding studies investigating biological effects of contaminants in the marine environment and to develop tools to be applied in marine environmental monitoring programmes. This issue is of particular relevance for national and international regulatory Commissions assessing environmental impacts of offshore oil and gas industries.

Source of the information presented

The 2003 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.

Summary

The ACME took note of the progress made in the ICES Sea-Going Workshop on Biological Effects of Contaminants in Pelagic Ecosystems (BECPELAG). The project originated in WGBEC and detailed planning was subsequently carried out by the ICES Steering Group for a Sea-going Workshop on Pelagic Biological Effects Methods (SGSEA). The major BECPELAG objectives were:

1) to assess the ability of selected methods to detect biological effects of contaminants in pelagic ecosystems;
2) based on the results, to recommend methods suitable for future monitoring programmes on biological effects of contaminants.

A detailed overview of the BECPELAG objectives and its components, as well as its status, was made at the 2001 and 2002 ACME meetings (ICES, 2001, 2003). More information about the workshop can be found at the BECPELAG website (http://www.niva.no/pelagic/web/) and a brief overview is attached as Annex 1.

The practical work was carried out in 2001 at four stations each on contaminant gradients in two areas: the German Bight and the Statfjord offshore oil industry area. It involved field sampling of pelagic organisms, deployment of cages for exposure experiments, and the collection of water samples and extracts for subsequent bioassays and chemical analyses. The analysis of the samples was finalized in 2002. The data generated within the different BECPELAG components were submitted to the BECPELAG database established at the University of Bremen, Germany, for a holistic statistical analysis, aiming at an integrated assessment of the biological and chemical data. This analysis has not yet been finalized.

Results from the different methods applied were presented at the BECPELAG wrap-up conference hosted by the European Environment Agency (EEA) in Copenhagen, 19–21 August 2002. Selected and aggregated results were also presented during a Theme Session at the 2002 ICES Annual Science Conference. Single scientific contributions by workshop participants as well as the results of the holistic data assessment will be published in a special peer-reviewed volume of the Society for Environmental Toxicology and Chemistry (SETAC), expected to be published by the end of 2003.

Some of the intermediate results of the BECPELAG Workshop are provided below in the section "Scientific background". Based on the results so far available, the following conclusions addressing the design of monitoring activities to assess biological effects of contaminants in pelagic ecosystems have been drawn and are endorsed by ACME:

1) The selection of methods depends on the objective, whether the focus is on local/regional effects monitoring, general effects monitoring in a wider area, or on the identification of specific substances and whether ecosystem relevance is important;
2) Monitoring and research programmes should include chemistry, biology, and modelling components (they are not mutually exclusive);
3) *In situ* extraction combined with bioassays, caging, and field sampling are complementary approaches;
4) *In situ* extracts are useful to identify mechanisms of toxicity and specific substances, caging provides a direct link to local exposure, whereas field sampling is considered more relevant in ecological terms.

Recommendations and advice

ICES recommends that Member Countries and the international regulatory Commissions take note of the progress made in the ICES Workshop on Biological Effects of Contaminants in Pelagic Ecosystems (BECPELAG). It is emphasized that the results achieved and conclusions drawn are of considerable relevance for the design of monitoring programmes aiming at an assessment of the environmental impact of offshore oil and gas industries.

Selected results of the ICES BECPELAG Workshop

Modelling

Hydrographical modelling was done for both study areas. Whereas the exposure situation appears to have been stable at locations in the Statfjord transect (at least during the cage deployment), the current regime was more variable at the German Bight locations. Especially the outer stations in the transect appeared to have been affected by different water masses. The main exposure at the German Bight stations would nevertheless be from the rivers Elbe and Weser.

Chemical analyses

A wide range of contaminants was analysed in the BECPELAG chemistry programme. The main focus was on biological matrices, i.e., fish (herring, saithe, and mackerel), blue mussel, caged cod and zooplankton, but extracts (semi-permeable membrane devices (SPMDs), Diffusive Gradient in Thin Film (DGTs), produced water) were also analysed for, e.g., PAHs, alkylphenols, and metals.

PAH concentrations in caged mussels decreased along both transects, but overall concentrations were higher in the oil-rig transect (Figure 4.1.1.1). The slightly elevated concentration at station G4 in the German Bight transect is probably related to the proximity of a Danish gas field.

In addition to PAHs, PCB concentrations in caged mussels also decreased from G1 to G4 on the German Bight transect. Perhaps surprisingly, there were no obvious trends for contaminants in the livers of caged cod. Also somewhat surprisingly, there was a significant decrease in concentrations of organotin compounds in mackerel liver along the German Bight transect. Similar results were seen for herring liver, but sample sizes were too small for clear conclusions. PAH concentrations in zooplankton samples from the Statfjord area were highest close to the platform, but otherwise levels varied with distance.

SPMDs accumulated organic contaminants in both areas and the highest concentrations of PAHs were found at the stations closest to the expected source (S1 and G1). There was, however, no clear gradient for total organic contaminants in either of the two areas. Alkylated PAHs dominated in SPMDs near Statfjord, but were not found in the German Bight. Alkylphenols were detected in SPMDs deployed in the German Bight, but not in SPMDs from Statfjord. The extraction method was not optimized for alkylphenols, so those results must be interpreted with caution.

Field-collected organisms

Field-collected organisms studied ranged from bacteria to fish. Bacterial diversity was assessed at some sites, but any site-dependent differences have not yet been quantified. Although there was apparent reduced grazing capacity by microzooplankton at one location in the German Bight, there was no obvious link to exposure. No effects were observed on primary production by algae cultured on board the vessel. Although there is an extensive database on fish embryo aberrations from the German Bight, there is only limited background from the northern North Sea. There were no obvious increases in fish larvae aberrations along any of the transects. There were no differences in PAH metabolite levels in bile from saithe (Statfjord transect only), herring (mainly German Bight transect) or mackerel collected at different sites. Although there was no difference between stations in EROD (cytochrome P4501A) activity, concentrations of cytochrome P4501A in herring liver differed between sites in the German Bight. DNA adducts were found in fish larvae (sandeel) in the German Bight, but there were no differences between locations. One interesting finding

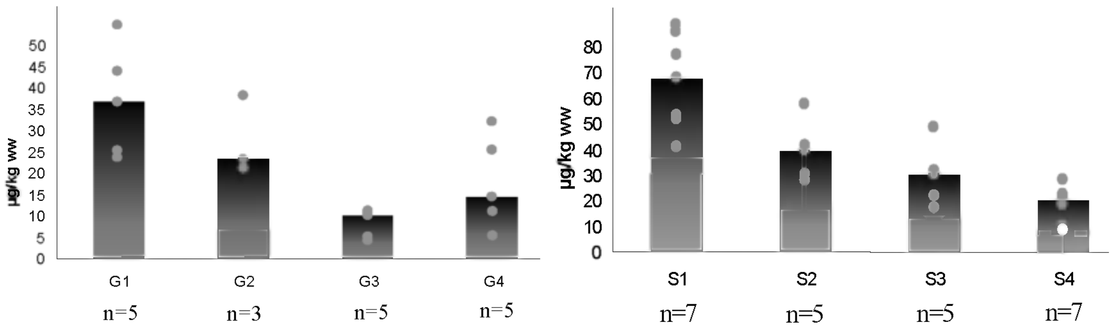

Figure 4.1.1.1. Sum of PAHs in blue mussels caged at the indicated locations; median and individual values shown. German Bight – left panel, Statfjord – right panel. G1 is closest to the Weser/Elbe estuaries, S1 closest to the oil platform.

was the differences between sites in liver tissue integrity (histopathology). Significant effects on herring and saithe liver tissue integrity were found in the inner German Bight (G1) and close to the Statfjord B platform (S1). There were no significant differences in plasma vitellogenin levels in either species in either area.

Exposure experiments

Various analyses were conducted on caged blue mussels and Atlantic cod. Clear differences along the transects were seen for histopathological changes in the blue mussel hepatopancreas (basophilic cell volume) and there was also reduced lysosomal stability at stations in the inner German Bight and close to the oil rig. In the German Bight, acetylcholinesterase (AChE) was inhibited in both blue mussels and cod. Further, benzo[*a*]pyrene hydroxylase (BaPH) was induced in blue mussels kept in the inner German Bight. There were no significant changes in scope for growth of mussels kept at the different sites in either area. The metal-responsive protein metallothionein also did not differ between sites. Tissues from blue mussels were subjected to two-dimensional electrophoresis. Protein patterns differed between sites close to contaminant sources and the reference sites. It is still too early to conclude on the utility of this method, but the relevant proteins should be identified.

There were obvious gradients in concentrations of PAH metabolites in both areas. In the Statfjord transect, alkylated PAH metabolites comprised a substantial fraction not seen in bile from cod kept in the German Bight. In the German Bight, but not in the Statfjord transect, there was a gradient in EROD activity. Glutathione-*S*-transferase (GST) activity increased at the sites closest to the oil rig.

There were low concentrations of DNA adducts in cod from both areas and no significant differences in vitellogenin levels (although there appeared to be a gradient). In the German Bight, there were differences between sites in cellular energy allocation (CEA).

Bioassays

The work on bioassays was hampered by contamination of extracts with triolein from the SPMDs, which had to be cleaned before analyses. Presumably due to the small amount of samples, no response was seen following intraperitoneal injection of extracts into juvenile salmon. Most of the methods applied only saw responses in the positive control, the produced water extract. This included all whole-organism tests, i.e., the ELS zebrafish test, mussel embryos, *Acartia*, *Tisbe*, and *Skeletonema*. In addition, responses were only seen in the produced water extract for the Comet assay (cell line) and microtox/mutatox. Some methods did, however, indicate effects in other extracts (closer to contaminant sources), e.g., fish primary hepatocyte culture (EROD, vitellogenin) and DR-CALUX. A preliminary TIE (toxicity identification and evaluation) identified oestrogenicity due to C3–C9 phenols in the produced water extract. Although conditions were optimal for the collection of sea surface microlayer (SSML) samples, effects were only seen in a few samples. Phototoxicity (exposure with UV) was evident in a single sample.

Some of the results indicate a possible interaction between 2–3 ring PAHs and the larger, more carcinogenic PAHs at the Statfjord transect. Produced water effluents are special in that they contain a very large proportion of low-molecular-weight PAHs compared to heavier PAHs. The results indicate that the lighter PAHs may inhibit the cytochrome P4501A system, thereby decreasing the development of DNA adducts. Although found in some earlier studies, the results need to be confirmed with experimental studies using appropriate PAH mixtures.

Data treatment

Data generated within various components of the BECPELAG Workshop were submitted by the participants to the BECPELAG database established at the University of Bremen, Germany, to be subject to a holistic integrated statistical analysis. The data assessment has not yet been completely finalized. Methods applied include:

1) generalized linear models (univariate response) and partial least squares (multivariate response) for exploring the relationship between biological effects and explanatory variables;
2) regression and classification trees for explanatory analysis and for recommending a battery of test biomarkers;
3) multivariate methods (e.g., principal component analysis, factor analysis) for constructing indices of biomarkers and contaminants.

It is envisaged that the BECPELAG data will be made available to the ICES Marine Data Centre, after approval by the data originators, for further assessments.

Acknowledgements

The ACME appreciated that the ICES BECPELAG Workshop has accomplished its aims and emphasized that guidance for offshore pelagic monitoring derived from the BECPELAG Workshop has already been adopted by Norwegian authorities and the offshore industry.

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ICES. 2003. Report of the ICES Advisory Committee on the Marine Environment, 2002. ICES Cooperative Research Report, 256: 6–7.

4.1.2 Tables for Recommended and Promising Methods for Biological Effects Monitoring

Request

This is an update of material presented in the 1999 ACME report and is part of the continuing ICES work to review and evaluate marine biological effects monitoring techniques in relation to programmes in Member Countries and the OSPAR Coordinated Environmental Monitoring Programme (CEMP).

Source of the information presented

The 2003 report of the Working Group on Biological Effects of Contaminants (WGBEC) and ACME deliberations.

Scientific background

Revision of criteria to define recommended techniques for monitoring

In order to improve the biological effects tools for monitoring purposes, ICES regularly updates information on recommended and promising methods for use in national and international monitoring programmes. Based on the criteria developed by WGBEC in 1994 and 1995, ICES provided lists of recommended and promising methods in 1997 (ICES, 1997) and a first update in 1999 (ICES, 1999).

The criteria for recommended methods were defined as:

1) methods should have been adequately tested in both the laboratory and the field;
2) satisfactory dose-response or concentration-response relationships should be known for a reasonable number of chemicals;
3) they should be practical methods that have been approved by the WGBEC. It is desirable that as many methods as possible should be published in the *ICES Techniques in Marine Environmental Sciences* (TIMES) series;
4) their sensitivity compared with existing equivalent monitoring techniques should be known, if appropriate.

Following discussions on the current relevance of the criteria at the WGBEC meeting in 2002 and subsequent ACME deliberations in the same year, the decision was made that the above criteria required revision. The main reason for the revision was that it is often difficult to establish direct cause and effect relationships between contaminants and effects and, as such, criterion 2) may not be satisfied, even though biological effects are observed in both the laboratory and the field. This is especially the case for fish diseases and benthic community structures, which are integrated endpoints of both anthropogenic and natural influences on the environment. The problem is, however, also relevant for many other intermediate biological effects processes.

The ACME agreed on the following revised criteria for recommended methods for monitoring programmes:

1) A recommended method needs to be an established technique that is available as a published method in the TIMES series or elsewhere.
2) A recommended method (or combination of methods) should have been shown to respond to contaminant(s) exposure in the field.
3) A recommended method (or combination of methods) should be able to differentiate the effects of contaminants from natural background variability.

Over the last few years, the availability of adequate quality assurance/analytical quality control (QA/AQC) procedures has become an increasingly important prerequisite for the implementation of the techniques in international monitoring programmes such as the OSPAR CEMP. These procedures comprise specific intercalibration exercises and, in recent years, these exercises have been developed for several established biological effects tools within the QUASIMEME and BEQUALM programmes. The ACME recognized that further developments are needed for those recommended techniques for which quality management tools are not yet in place.

Review of tables for recommended and promising methods for biological effects monitoring

Based on the 2003 criteria for the evaluation of the status of monitoring techniques to assess contaminant effects in marine ecosystems, WGBEC updated the existing lists of "recommended" and "promising" techniques.

The ACME noted that the structure and format of the tables had changed since the last update in 1999 (ICES, 1999). Further, recommended methods for fish and invertebrates have been separated into two tables and the status of intercalibration is clearly indicated. The revised lists for recommended methods for fish and invertebrates are given in Tables 4.1.2.1a and 4.1.2.1b. The revised list for promising methods is contained in Table 4.1.2.2 and methods for future consideration are listed in Table 4.2.2.3.

Three methods (Yeast Estrogenic Screen (YES), Yeast Androgenic Screen (YAS), and embryo aberrations and reproductive disorders in field-collected amphipod crustaceans) are new in the list of promising techniques requiring further research.

Table 4.1.2.1a. Recommended techniques for biological monitoring programmes at the national or international level – methods for fish.

Method	Organism	Intercali-bration[1]	Issues addressed	Biological significance	References
Bulky DNA adduct formation	Fish	B	PAHs; Other synthetic organics, e.g., nitro-organics, amino triazine pesticides (triazines)	Measures genotoxic effects. Possible predictor of pathology through mechanistic links. Sensitive indicator of past and present exposure.	1–6
AChE inhibition	Fish	O	Organophosphates and carbamates or similar molecules. Possibly algal toxins	Measures exposure.	7–10
Metallothionein induction	Fish	B	Measures induction of metallothionein protein by certain metals (e.g., Zn, Cu, Cd, Hg)	Measures exposure and disturbance of copper and zinc metabolism.	11–15
EROD or P4501A induction	Fish	B	Measures induction of enzymes which metabolize planar organic contaminants (e.g., PAHs, planar PCBs, dioxins)	Possible predictor of pathology through mechanistic links. Sensitive indicator of past and present exposure.	16–22
ALA-D inhibition	Fish	B	Lead	Index of exposure.	23–24
PAH bile metabolites	Fish	Q	PAHs	Measures exposure to and metabolism of PAHs.	25–26
Lysosomal stability	Fish		Not contaminant-specific, but responds to a wide variety of xenobiotic contaminants and metals	Measures cellular damage and is a good predictor of pathology. Provides a link between exposure and pathological endpoints. Possibly, a tool for immunosuppression studies in white blood cells.	27–29
Early toxicopathic lesions, pre-neoplastic and neoplastic liver histopathology	Fish	B(?)	PAHs; Other synthetic organics, e.g., nitro-organics, amino triazine pesticides (triazines)	Measures pathological changes associated with exposure to genotoxic and non-genotoxic carcinogens.	30–38
Externally visible lesions and parasites	*Limanda limanda*, *Platichthys flesus*, *Gadus morhua*	B(?)	Responds to a wide variety of environmental contaminants and non-specific stressors	Integrative response; measures general fish health; elevated prevalence may indicate exposure to contaminants	39–42
Vitellogenin induction	Male and juvenile fish	B(?)	Oestrogenic substances	Measures feminization of male fish and reproductive impairment.	43–47
Intersex	Male fish		Oestrogenic substances	Measures feminization of male fish and reproductive impairment.	48–49
Reproductive success in *Zoarces viviparus*	*Zoarces viviparus*	B	Not contaminant-specific, will respond to a wide range of environmental contaminants	Measures reproductive output and survival of eggs and fry in relation to contaminants. Restricted to period when young are carried by female viviparous fish.	50

[1]B: Subject to BEQUALM; O: Other, e.g., nationally coordinated exercises or exercises within EU projects; Q: Subject to QUASIMEME.

Table 4.1.2.1b. Recommended techniques for biological monitoring programmes at the national or international level – methods for invertebrates.

Method	Organism	Intercalibration[1]	Issues addressed	Biological significance	References
AChE inhibition	Bivalve molluscs	O	Organophosphates and carbamates or similar molecules. Possibly algal toxins	Measures exposure.	51
Metallothionein induction	*Mytilus* spp.	O	Measures induction of metallothionein protein by certain metals (e.g., Zn, Cu, Cd, Hg)	Measures exposure and disturbance of copper and zinc metabolism.	52–53
Lysosomal stability	*Mytilus* spp.	O	Not contaminant-specific, but responds to a wide variety of xenobiotic contaminants and metals	Measures cellular damage and is a good predictor of pathology. Provides a link between exposure and pathological endpoints. Possibly, a tool for immunosuppression studies in white blood cells.	54
Lysosomal neutral red retention	*Mytilus* spp. Oyster	B	As for lysosomal stability.	As for lysosomal stability.	55
Whole sediment bioassays	*Corophium* *Echinocardium* *Arenicola* *Leptocheirus* *Grandidierella* *Rhepoxynius* *Ampelisca*	B	Not contaminant-specific, will respond to a wide range of environmental contaminants in sediments	Acute/lethal and acute/sub-lethal toxicity only at present. May enable retrospective interpretation of community changes.	56–60
Bioassays of sediment pore waters, sea water elutriates, sea water samples	Water column organisms including: • *Dinophilus* larvae • sea urchin embryo • bivalve embryo • *Acartia* • Microtox		Will respond to a wide range of environmental contaminants. Useful for dredge spoils, sediments liable to resuspension	Acute and sub-lethal toxicity, including genotoxicity, etc. Toxicity of hydrophobic contaminants might be underestimated in pore water assays.	61–66
Scope for growth	Bivalve molluscs, e.g., *Mytilus* spp. and oysters	O	Responds to a wide variety of contaminants	Integrative response; a sensitive sub-lethal measure of energy available for growth.	67–70
Shell thickening*	*Crassostrea gigas*		Specific to organotins	Disruption to pattern of shell growth.	71
Imposex	Neogastropod molluscs, e.g., dogwhelk (*Nucella lapillus, Buccinum undatum*)	Q	Specific to organotins	Reproductive interference. Estuarine and coastal littoral waters (*Nucella*) and offshore waters (*Buccinum*).	72–74
Intersex	*Littorina littorea*	Q	Specific to reproductive effects of organotins	Reproductive interference in coastal (littoral) waters.	75–76
Benthic community analysis	Macro-, meio-, and epibenthos	B	Responds to a wide variety of contaminants, particularly those resulting in organic enrichment	Ecosystem level. Retrospective. Particularly useful for point sources. Most appropriate for deployment when other monitoring methods indicate a problem may exist.	77–82

[1]B: Subject to BEQUALM; O: Other, e.g., nationally coordinated exercises or exercises within EU projects; Q: Subject to QUASIMEME.

*Removed from OSPAR Guidelines on TBT-specific biological effects.

Table 4.1.2.2. Promising biological effects monitoring methods that require further research before they can be recommended for monitoring (both fish and invertebrates).

Method	**Organism**	**Issue addressed**	**Biological significance**	**References**
H6PD/G6PD enzyme-altered foci	Fish	PAHs; Other synthetic organics, e.g., nitro-organics, amino triazine pesticides (triazines)	Indicates exposure to carcinogen(s). Index of chronic exposure.	83–90
DNA strand breaks including Comet assay	Fish, mussels, cells	Not contaminant-specific, will respond to a wide range of environmental contaminants	Measures genotoxic effects, but is also extremely sensitive to other environmental parameters.	91–92
Apoptosis	Fish cells	Responds to a wide range of contaminants	General response.	93
AChE inhibition	Invertebrates	Organophosphates and carbamates or similar molecules. Possibly algal toxins	Measures exposure.	11, 94–95
Oncogenes	Fish	PAHs; Other synthetic organics, e.g., nitro-organics, amino triazine pesticides (triazines)	Activation of oncogenes (*ras*) or damage to tumour suppressor genes (p53). Measures genotoxic effects leading to carcinogenesis.	96–98
P4501A induction	Invertebrates	Induced enzyme response to PAHs, planar PCBs, dioxins and/or furans	Measures exposure to organic contaminants.	99–100
Glutathione-*S*-transferase(s) (GST)	Fish, mussels	Predominantly organic xenobiotics	Measures exposure and the capacity of the major group of phase II enzymes.	101–102
Multi-drug/multi-xenobiotic resistance (MDR/MXR)	Fish, invertebrates	Organic xenobiotics	Measures an adaptation to xenobiotic stress.	103–114
Oxidative stress	Fish *Mytilus* spp.	Not contaminant-specific, will respond to a wide range of environmental contaminants	Measures the presence of free radicals.	115–117
Immunocompetence	Fish, invertebrates	Not contaminant-specific, will respond to a wide range of environmental contaminants	Measures factors that influence susceptibility to disease.	118
On-line monitoring	Mussels and crabs	Not contaminant-specific, will respond to a wide range of environmental contaminants	Measures the effects of chemicals on heart rate using a simple and inexpensive remote biosensor. Gives an integrated response.	119
Abnormalities in wild fish embryos and larvae	Fish, including demersal and pelagic species	Not linked unequivocally to contaminants	Measures frequency of probably lethal abnormalities in fish larvae. Mutagenic, teratogenic.	120–121
Chronic whole sediment bioassays	Invertebrates	Responds to a wide range of contaminants	Measurements such as growth and reproduction, coupled to biomarker responses, which will give a measure of the bioavailability and chronic toxicity in whole sediments.	45
Pollution-induced community tolerance (PICT) water bioassay	Microalgae, bacteria	Specific contaminants can be tested	Measure of degree of adaptation to specific pollutants. Not yet widely tested; retrospective.	122–124

Table 4.1.2.2. Continued.

Method	Organism	Issue addressed	Biological significance	References
ELISA for DNA adducts	Fish	Not contaminant-specific	Genotoxic effects.	125–127
DR-CALUX	Reporter gene assay	Ah receptor-active compounds	Possible predictor of pathology.	128
ER-CALUX	Reporter gene assay	Oestrogen receptor-active compounds	Potential endocrine disruption.	129
Bulky DNA adduct formation	Mussels, invertebrates	PAHs; other synthetic organics	Measures genotoxic effects.	130–133
Gene arrays	Fish	Various	Combined responses from various biomarkers.	134
Histopathology	Invertebrates	Not contaminant-specific	General responses.	135–138
YES	Reporter gene assay (yeast)	Oestrogen receptor-active compounds	Potential endocrine disruption.	139–140
YAS	Reporter gene assay (yeast)	Androgen receptor-active compounds	Potential endocrine disruption.	141–142
Embryo aberrations, reproduction disorders in field-collected amphipod crustaceans	All amphipod species	Contaminant-specific	Measures frequency of different types of lethal embryo aberrations; allows for separating effects of contaminants and environmental climate variables.	143–147
GSI	Fish	Not contaminant-specific	General response.	

Notes: ELISA: Enzyme-linked immunosorbent assay; DR-CALUX: dioxin-responsive chemical-activated luciferase gene assay; ER-CALUX: oestrogen-responsive chemical-activated luciferase gene assay; YES: Yeast oestrogen screen; YAS: Yeast androgen screen; GSI: gonado-somatic index.

Table 4.1.2.3. Methods to be considered in more detail at the 2004 WGBEC meeting.

Method	Organism	Issue addressed	Biological significance
Spiggin	Three-spined stickleback	Androgens	Measures environmental androgens
Micronuclei	Fish, bivalve molluscs	Not contaminant-specific	General response
Aromatase induction/inhibition	Fish	Not contaminant-specific	Potential reproductive toxicology
Delayed reproduction	Fish	Not contaminant-specific	Reproductive disruption

Recommendations and advice

ICES recommends that OSPAR consider the adoption of recommended techniques that have adequate quality management tools in place for implementation in their monitoring programmes, with the precaution that for some recommended tools developments of quality management tools are currently in progress or should be initiated, as is indicated in the tables.

ICES recommends that Member Countries initiate the development of quality management tools for recommended techniques for which these developments have not yet started.

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4.1.3 Use of Toxicity Identification and Evaluation (TIE) Procedures in Monitoring Programmes

Request

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

Source of the information presented

The 2003 report of the Working Group on Biological Effects of Contaminants (WGBEC) and ACME deliberations.

Scientific background

The use of Toxicity Identification and Evaluation (TIE) as a strategy for effect-based marine monitoring to assess the quality of the marine environment, integrating effect measurements and identification of the chemicals causing the observed effects, was discussed previously by the ACME in 2001 (ICES, 2001). At that time, the ACME noted the information with great interest and confirmed that the technique would form a basis for future monitoring.

In 2003, WGBEC updated information on recent developments in this field. As phase I TIE usually follows the classical approaches described in, e.g., the U.S. Environmental Protection Agency (EPA) guideline documents (USEPA, 1991, 1992, 1996), a new element in the phase II TIE approach is that fractionation and identification procedures are based on the most recent developments of bioassays to simplify the sample matrix, followed by broad-scan high-resolution chemical analysis, such as high-performance liquid chromatography (HPLC), gas chromatography-mass spectrometry (GC-MS), Automated Mass Spectral Deconvolution System (ADMIS), Quality Peak Identification and Database system (QPID), and QSAR[1] models. A key element in TIE is the use of small-scale bioassay methods: the oyster embryo bioassay, copepod (*Tisbe battagliai*), Microtox®, Mutatox®, microalgae, the YES[2] and YAS[3] assays, DR-CALUX[4], and fish hepatocytes. This technique is also known as bioassay-directed fractionation (BDF).

Several countries (UK, Germany, Norway, Sweden, and The Netherlands) currently apply the technique in case studies on a regular basis. As an example, the Centre for Environment, Fisheries and Aquaculture Science (CEFAS) (UK) has used several of the TIE procedures in a number of studies, including:

- Direct toxicity assessment (DTA) of complex effluents;
- Identifying toxicants in estuarine surface waters where positive bivalve embryo results had been measured;
- Identifying the cause of a fish kill, the results of which led to identifying a method of water treatment;
- Characterizing the toxic component in pharmaceutical waste;
- Identifying toxicants in a headwater stream;

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[1]QSAR: quantitative structure-activity relationships.

[2]YES: yeast oestrogen screen; genetically modified yeast cells that respond to oestrogens.

[3]YAS: yeast androgen screen; genetically modified yeast cells that respond to androgens.

[4]DR-CALUX: Dioxin-Responsive chemical-activated LUciferase Expression (CALUX®) Assay.

- Characterizing the toxicity of marine sediments;
- Identifying oestrogenic compounds in surface waters;
- Identifying androgenic compounds in surface waters;
- Identifying oestrogenic compounds in offshore produced waters.

The currently available knowledge permits the following conclusions:

1) TIE/BDF is the most appropriate technique for the identification of the substances responsible for measurable biological effects;
2) TIE/BDF techniques have shown that unknown biologically active substances are present in the environment. It is unknown whether these are of natural or anthropogenic origin;
3) Even though TIE techniques have provided many answers, there is still an increasing need to identify the remaining unidentified compounds through advanced chemical analyses;
4) Over the past decade, significant advances have been made in the development of the tools available to a point where the approach is globally accepted. However, depending on why a TIE study has been initiated, the identification of individual compounds may not be necessary if remediation/toxicity reduction options can be found;
5) Certain countries use (e.g., UK) or intend to use (e.g., The Netherlands) their TIE procedures to inform their regulatory and monitoring programmes in the near future;
6) A new use of this technique is the development of methods for the characterization of hazardous substances in tissues or body fluids, e.g., bile.

The ACME concluded that TIE procedures provide a useful tool for characterizing biologically active compounds in environmental samples (water, effluent or sediments), particularly if modified to the requirements of an individual study. The procedures are scientifically sound and the approach has global acceptance.

The ACME supports further research and development on substance isolation and identification.

Recommendations and advice

TIE is an important strategy for the integration of biological and chemical methods in monitoring as a problem-solving tool. There is a need for quality assurance/quality control (QA/QC) procedures for TIE techniques if they are to be used for regulatory purposes or monitoring (possibly in the form of reference material provided through BEQUALM). ICES recommends the application of TIE techniques for relevant environmental problems to gain more experience in its use in wider applications.

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4.2 Guidelines for Integrated Chemical and Biological Effects Monitoring

Request

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

Source of the information presented

The 2003 reports of the Marine Chemistry Working Group (MCWG), the Working Group on Biological Effects of Contaminants (WGBEC), and the Working Group on Marine Sediments in Relation to Pollution (WGMS), and ACME deliberations.

Scientific background

The underlying objective of this approach is to develop a methodology with which to evaluate the health of marine ecosystems. There is general agreement that the best way to achieve this objective is by using a suite of chemical and biological analyses in an integrated fashion. Monitoring in the past has been based mostly on chemical measurements, but biological effects techniques have become increasingly important over recent years. As such, they have become part of the OSPAR Joint Assessment and Monitoring Programme (JAMP) and, more recently, also of the OSPAR Coordinated Environmental Monitoring Programme (CEMP).

The development of guidelines for integrated chemical and biological effects monitoring will require a thorough review of recent activities in this field as well as more integrated work among the three ICES Working Groups involved. Each of these Working Groups will consider this issue at their 2004 meetings.

The OSPAR JAMP guidelines recommend different types of biological effects monitoring tools for different objectives. There are guidelines for General Quality Assessment, Local Impact Assessment, and Contaminant-specific Monitoring. This strategy was formulated several years ago and is currently under revision. Under the OSPAR JAMP, the mandatory monitoring programme CEMP is implemented, which includes imposex measurements beginning in 2003 and will include other biological effects measurements when monitoring guidelines and QA protocols are complete.

In view of these developments, WGBEC reviewed the present status of the use and integration of biological effects methods in currently ongoing national monitoring programmes with the objectives to:

- generate an overview of the implementation of biological effects techniques in the national programmes;
- check the level of integration of the techniques in an integrated multidisciplinary programme;
- estimate the compliance of the programmes with the CEMP; and
- evaluate progress with QA activities within Member Countries.

Information on national monitoring activities was received from Belgium, Canada, France, Germany, Iceland, Norway, Sweden, The Netherlands, and the UK.

The ACME noted that all of these countries have started to incorporate, or have already incorporated, biological effects methods in their monitoring programmes in an increasingly integrative way. The biological effects monitoring in the OSPAR countries is, to a large extent, related to several JAMP products and comprises biomarker, bioassay, and biological community measurements, and monitoring organizations are increasingly in agreement with the QA/AQC requirements.

Several agencies and projects with an interest in QA/AQC of biological effects measurements were identified, including BEQUALM, BEEP, and OECD, and it was felt that the links between these organizations should be consolidated.

In general, chemical and biological effects techniques can be combined at three levels:

1) Simultaneous determinations of an appropriate set of chemical determinands and biological responses in the same organisms. In addition, other health-related parameters need to be determined (e.g., condition, size) as well as supporting environmental parameters (e.g., organic carbon) and environmental conditions (e.g., salinity, temperature).
2) Bioassay-directed fractionation of environmental matrices, followed by high-resolution chemical analyses to determine the substances causing effects (the Toxicity Identification and Evaluation (TIE) approach) (see Section 4.1.3, above).
3) Caging experiments, where chemical and biological effects parameters are determined on a specific set of organisms together with supporting parameters.

Chemical measurements are generally made for those compounds that are a known threat to the environment and with the specific aim of determining their spatial distribution and temporal trends. It needs to be clarified whether sampling for these purposes is compatible with biological effects measurements. For instance, the CEMP requires both the chemical determination of PAHs in sediment and biota (both for spatial distribution and temporal trend monitoring) and the determination of PAH-specific biological effects. The CEMP could therefore be one platform within which to develop integrated monitoring. This will require input from experts in the field of chemical analysis and biological effects techniques, as well as statisticians. It will require not only adequate sampling to achieve both goals, but also an agreement on the methodology, specifically for the biological effects measurements. Guidelines on integration may prove to be possible only for specific cases such as the CEMP, but a general strategy towards integration can also be developed. WGBEC suggested that the most effective way to proceed would be to organize a joint OSPAR/ICES workshop to prepare such guidelines.

Recommendations and advice

The ACME concluded that a workshop should be organized as the best means to develop final guidelines for integrated biological and chemical monitoring. Terms of reference for this workshop include to:

a) develop specific guidelines for the integration of chemical and biological effects techniques with special emphasis on those that have become mandatory in the CEMP;
b) develop guidelines towards integrated chemical and biological effects monitoring for the entire range of issues in the JAMP;
c) develop threshold values for integrated biological effects to indicate action levels which could initiate remedial measures.

Participants in the workshop should include selected experts from the three working groups already involved (MCWG, WGBEC, WGMS), as well as from the ICES Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) and the Working Group on Statistical Aspects of Environmental Monitoring (WGSAEM) in addition to experts from OSPAR. The timing of the workshop will need to be considered in relation to the other activities already being scheduled in relation to the JAMP, as well as the need to provide for adequate preparations in advance of the workshop.

4.3 Procedures for Temporal Trend Monitoring of Contaminants in Sediments

Request

This is part of continuing ICES work to provide a basis for scientific advice on current needs in relation to the monitoring and assessment of temporal changes in sediment quality within the ICES area.

Source of the information presented

The 2003 report of the Working Group on Marine Sediments in Relation to Pollution (WGMS) and ACME deliberations.

Scientific background

In 2002, the ACME noted the strong relevance of obtaining a comprehensive, accurate inventory of strategies and procedures for monitoring contaminants in sediments used by ICES Member Countries, available alongside the sediment data, in relation to temporal trend assessment purposes. Furthermore, the ACME considered it very important to take sediment dynamics into account when trend studies are planned (ICES, 2003). The ICES area is very large and a number of countries have long coastlines, which include areas with different sediment dynamics.

Work on the inventory of national procedures for sediment trend monitoring was finalized by WGMS in spring 2003. Intersessionally, the ICES Marine Data Centre had contributed with an overview of sediment data held in the databases at ICES. This overview has been incorporated in the inventory. The inventory contains a short explanation of the various temporal trend strategies available, followed by a country-by-country description of planned monitoring programmes for the coming years.

The ACME reviewed this document and agreed that it contained useful information in relation to monitoring temporal trends of contaminants in sediments. The final document is attached as Annex 2. It will also be incorporated as an annex to the ICES Sediment Monitoring Guidelines.

The ACME noted that work to develop guidance on the interpretation of trend monitoring data on contaminants in sediments, taking into account sediment dynamics, is progressing in WGMS. WGMS foresees the submission of a final document on the issue of sediment dynamics to ACME at its 2004 meeting.

Recommendations and advice

ICES recommends that Annex 2, the Inventory of National Programmes for Trend Monitoring of Contaminants in Sediments, be considered by OSPAR and HELCOM for use in their trend assessments of contaminants in sediments.

ICES also recommends that Member Countries submit data sets on contaminants in marine sediments to the ICES Marine Data Centre, for use in the assessment of temporal trend data on contaminants in sediments and biota coordinated by OSPAR in 2004.

Reference

ICES. 2003. Report of the ICES Advisory Committee on the Marine Environment, 2002. ICES Cooperative Research Report, 256: 12–14.

4.4 Preferred Matrix for Monitoring Priority Compounds under the Water Framework Directive

Request

This is new work to follow the developments under the EU Water Framework Directive with the aim of promoting harmonization of approaches.

Source of the information presented

The 2003 report of the Marine Chemistry Working Group (MCWG) and ACME deliberations.

Scientific background

At the 2002 meeting of the Marine Chemistry Working Group (MCWG), the monitoring framework being developed by the EU Working Group on Analysis and Monitoring of Priority Substances and Pollution Control (AMPS) for implementation of the European Water Framework Directive (WFD) was discussed, and it was agreed that MCWG would seek to establish closer liaison with AMPS regarding appropriate methodology and related topics.

Following the MCWG meeting and at the direction of the ICES General Secretary, the Chair of MCWG contacted the Chair of AMPS. In January 2003, the Chair of MCWG attended the second meeting of the AMPS

Group held at the EU Joint Research Centre in Ispra, Italy, and gave a presentation on the work of the MCWG. It was agreed at that meeting that closer contact was desirable, and that one scientist who is a member of both AMPS and MCWG would act as the liaison member. The MCWG Chair then made a presentation at the 2003 meeting of MCWG on the work of the AMPS Group. The first two meetings of the AMPS Group have produced three major documents: 1) a table of existing standard methods for the analysis of priority substances; 2) a background document on analytical determination of substance groups (the polybrominated diphenylethers (PBDEs), the short-chain chlorinated paraffins (SCCPs), and alkylphenols); and 3) a table of available certified reference materials for priority substances. All of these documents were made available to MCWG.

To date, the AMPS Group has concluded that:

a) for about two-thirds of the priority substances, compliance with the proposed Environmental Quality Standards (EQS) can be checked using available standard methods;

b) for some fields of application, there are no standards available at present, e.g., for the analysis of priority substances in transitional and coastal waters, solid particulate matter, sediments, and biota;

c) for some of the priority substances, e.g., SCCPs (short-chain chlorinated paraffins), there are no standard methods available;

d) other validated procedures for the analysis of priority substances shall be taken into consideration as well as standard methods;

e) data quality requirements have to be identified;

f) there is no information on the costs of the monitoring of priority substances including the costs of sampling;

g) there is only limited information to show whether or not the listed standard methods have been widely applied in routine laboratories;

h) there is an urgent need for more certified standards and reference materials for QA/QC;

i) there is a need to establish a European laboratory proficiency testing scheme for inland water monitoring;

j) there is a need for further method development and validation studies for several priority substances.

Due to the high log K_{ow} value of many of the priority substances, monitoring in the water phase is not appropriate. Hence, other matrices must be taken into account for compliance monitoring, particularly suspended particulate matter (SPM). At present, SPM sampling is not established in most of the EU member states and accession countries, and SPM cannot be collected in all water body types to be monitored, e.g., SPM cannot be collected in clear-water lakes, or fast-flowing mountain rivers. Therefore, the identification of proper monitoring approaches for different water body types is under discussion.

During its meeting, MCWG developed a table of the priority substances under the WFD, indicating for which of these substances water would reflect the preferred matrix for monitoring purposes, and for which substances other matrices should be considered (sediment, SPM, biota), for submission to the third meeting of the AMPS Group, which was held in April 2003 (see Table 4.4.1, below).

The water matrix referred to in Table 4.4.1 is the dissolved phase. If the answer provided in the table is “no”, then other matrices should be considered (sediment, SPM, biota). These decisions should take account of the characteristics of the area to be monitored, as well as the aims of the monitoring programme. For instance, SPM concentrations may be very low in some areas due to geological variations, and some biota may rapidly biotransform the chemicals of interest so that their determination in tissues is not possible. In many cases, this may mean that sessile bivalve molluscs are preferred as sentinel species over fish.

The ACME agreed that the close liaison established between the AMPS Group and MCWG at working scientist level should continue, especially given that the monitoring scheme to be developed under the Water Framework Directive is likely to have a growing influence within other related monitoring programmes.

4.5 Global Ocean Observing System

Request

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

Source of the information presented

The 2003 reports of the ICES/IOC Steering Group on GOOS (SGGOOS), and the ICES/EuroGOOS Planning Group on the North Sea Pilot Project (PGNSP), and ACME deliberations.

Scientific background

The ICES/EuroGOOS Planning Group on the North Sea Pilot Project (PGNSP) discussed existing coupled physical-ecosystem models in nowcast mode within the planned NORSEPP (North Sea Pilot Project). Coupled physical-ecosystem modelling of the Northwest European shelf seas is under development in many Northwest European countries, and representatives of relevant agencies attended the meeting of PGNSP. There are a range of levels of complexity of ecosystem models,

Table 4.4.1. Preferred matrix for monitoring priority compounds under the Water Framework Directive.

Priority substance	Is water the preferred matrix ?
Alachlor	Yes
Anthracene	No
Atrazine	Yes
Benzene	Yes
Brominated diphenylethers	No
Cadmium and its compounds	No
C_{10}–C_{13} chloroalkanes	No
Chlorfenvinphos	Yes
1,2-Dichloroethane	Yes
Di(2-ethylhexyl)phthalate	No
Diuron	Yes
Endosulphan and alpha-endosulphan	Yes
Hexachlorobenzene	No
Hexachlorobutadiene	No
Hexachlorocyclohexane and lindane	Yes
Isoproturon	Yes
Lead and its compounds	No
Mercury and its compounds	No
Naphthalene	No
Nickel and its compounds	No
Nonylphenols and 4-nonylphenol	No
Octylphenols and *p-tert*-octylphenol	No
Pentachlorobenzene	No
Pentachlorophenol	Yes
Polycyclic aromatic hydrocarbons	No (considered individually, as a group)
Benzo[*a*]pyrene	No
Benzo[*b*]fluoranthene	No
Benzo[*k*]fluoranthene	No
Benzo[*ghi*]perylene	No
Fluoranthene	No
Indeno[1,2,3-*cd*]pyrene	No
Simazine	Yes
Tributyltin compounds and cation	No
Trichlorobenzenes (and 1,2,4-TCB)	Yes
Trichloromethane	Yes
Trifluralin	Yes

from the full complexity of the European Regional Seas Ecosystem Model (ERSEM), which includes both pelagic and benthic components, and a range of size groups for zooplankton, to the simpler phytoplankton-only models such as the Norwegian Ecological Model System (NORWECOM) running daily at the Norwegian Meteorological Institute.

In common with physical ocean forecast models, the coupled physical-ecosystem models in nowcast mode will take surface forcing as available from numerical weather prediction models. Thus, the ocean models are driven by "today's" weather. However, usually climatological values are used for river inputs.

Because of the complexity of coupled physical-ecosystem models and the required computing resource, there is usually a trade-off between the model domain and the model resolution. For research understanding, it has been shown that for the Irish Sea or the North Sea, a target resolution of around 1 nautical mile is needed, in order to resolve the baroclinic processes in the physical model, since they impact on the marine ecosystem.

The coupled models can be run for long-term hindcasts, where the necessary meteorological-ocean forcing and boundary data are available, or they can be run daily in nowcast mode. Usually the output files from the daily nowcast runs are kept, thus building up over time an archive database of model results.

Typical indicators produced by a model-based marine monitoring system include:

- plankton concentration;
- total, new or primary production;
- peak production of different algal groups;
- bottom oxygen concentrations;
- zoobenthos;
- oxygen consumption;
- nutrient concentrations and ratios;
- nutrient transports to target areas.

Model output can be presented in either a spatial view (GIS-compatible format) or as a time series representation at a selected location. Development of the best means of presenting modelled data to users is only just beginning, though some work is starting at a national level. This is a key step in the transition from research to operational modelling, and is part of the pre-operational development of the modelling systems.

Further development to pre-operational and operational status of coupled physical-ecosystem modelling will follow on from the work described above. In particular, planned developments to the ERSEM model will include dinoflagellates in the model system. For the physical modelling, in strongly tidal or shallow waters, the three-dimensional ocean circulation model needs to be coupled with a wave model, for improved representation of bed stress, sediment and nutrient resuspension, and surface stress. As computing resources increase, the spatial resolution of the shelf-wide coupled models can be increased, and also the complexity of the coupled model systems can be further increased.

The need for northwest shelf-wide hydrological modelling for improved representation of riverine inputs was noted, and some early developments are taking place at a national level. As coupled physical-ecosystem models become developed with the ability to nowcast phytoplankton and zooplankton, it will become possible in future to include Individual Based Models in the system.

The use of web-based "live access server" technology to access large gridded data sets of model output is already under development for data sets of physical ocean model output, and this could be extended to include output from coupled physical-ecosystem modelling. This software allows web-based access to the data, where the user can configure the presentation of the data, including selection of a sub-domain of the model (http://www.nerc-essc.ac.uk/las).

At the SGGOOS meeting, a comprehensive overview of the planning steps and current status of the ICES/EuroGOOS North Sea Ecosystem Pilot Project (NORSEPP) was given. National and international policy drivers behind NORSEPP have been reviewed and an illustrated definition of an operational (ocean) observing system (OOS) was provided using NORSEPP as the model. In addition, a framework for an ecosystem approach to resource management was proposed using the NORSEPP example, taking into consideration the current structure and function of ICES and its assessment working groups. SGGOOS concluded that, in order to accommodate an ecosystem-based management framework within ICES, the current structure and function of the various assessment working groups will have to be modified in the least and perhaps be totally revamped. A back-of-the-envelope calculation was made of the projected costs for running NORSEPP compared with the costs for the fisheries surveys and assessments; NORSEPP costs would represent about 2% of the fish assessment costs. The need for action soon on the implementation of NORSEPP was recognized and a plan was proposed. NORSEPP participants were identified and an agreement was made to produce a provisional status report on the North Sea for inclusion in next year's ICES Annual Ocean Climate Status Summary (see also Section 6.6.1).

SGGOOS discussed a plan for a workshop on variability in North Atlantic basic-scale circulation over recent decades and its forcing of (biological) variability in adjoining shelf ecosystems policy. The discussion focused on what approach SGGOOS should take to move the idea of a North Atlantic variability workshop forward. SGGOOS was reminded that the plan to develop a comprehensive and integrated Atlantic-wide ocean observation framework was discussed at the inception of SGGOOS, as an integral part of the SGGOOS Implementation Plan, which has been endorsed at all levels in GOOS. IOC has suggested a way forward based on the planning and implementation steps followed by the highly successful GODAE and ARGO projects. Members of SGGOOS felt that, at least in the short term, emphasis within the existing Study Group has to be placed on getting NORSEPP (and other coastal pilot projects such as GoMA-GOOS) off the ground. It was suggested that there will likely be need to capture an entirely new community (new people, resources) to move a North Atlantic initiative along in parallel with the GOOS regional pilot projects. It was agreed that IOC and the SGGOOS Co-Chairs would work intersessionally on drafting a strawman discussion paper to get the planning started.

5 QUALITY ASSURANCE PROCEDURES AND ACTIVITIES

5.1 Quality Assurance of Biological Measurements in the Baltic Sea

Request

Item 1 of the 2003 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 activities and ongoing ICES intercomparison exercises, and provide a full report on the results.

Source of the information presented

The 2003 report of the ICES/HELCOM Steering Group on Quality Assurance of Biological Measurements in the Baltic Sea (SGQAB) and ACME deliberations.

Summary

Great progress has been achieved in the development of the ICES Biological Data Reporting Format (BDRF). Simultaneously, poor submission of actual data to the ICES Marine Data Centre occurs. Slow progress is taking place in the development of taxonomic checklists for monitored marine organisms. A checklist for Baltic phytoplankton species has been finished and is available on the Algaline web page, and Integrated Taxonomic Information System (ITIS) codes are considered suitable for use. SGQAB revised relevant parts of the HELCOM COMBINE Manual and also discussed the QA Guidelines for measuring primary production and chlorophyll *a* to be included in the COMBINE Manual. It was decided to make further developments in the phytobenthos programme. The results were reviewed of the First German National Chlorophyll *a* Ring Test. The results obtained showed that all three methods compared are useful to determine the chlorophyll *a* content, but none of them is fit for the measurement of phaeopigments.

Recommendations and advice

ICES recommends to the countries of the HELCOM area and their institutes to continue active implementation and harmonization of QA procedures for biological monitoring.

ICES recommends that HELCOM consider the possibility of supporting future expert training courses and other QA-related activities, *inter alia,* by obtaining funding from the GEF-sponsored Baltic Sea Regional Project.

ICES recommends that HELCOM promote the development and publication of taxonomic checklists for monitored marine organisms and update the COMBINE Manual according to the suggestions of SGQAB, as accepted by ACME.

ICES recommends that Member Countries actively develop software that enables the transfer of national data to the ICES Biological Data Reporting Format and that these data be submitted to the ICES Marine Data Centre on a regular basis.

Scientific background

SGQAB reviewed the changes made to the ICES Biological Data Reporting Format (BDRF) and the status of data submissions. Great progress in the development of the BDRF was noted. However, only a few submissions of actual data have yet taken place.

During a joint session between SGQAB and the Steering Group on Quality Assurance of Biological Measurements in the Northeast Atlantic (SGQAE), information on the development of taxonomic checklists was presented. It was noted that several checklists are available on the websites of the Finnish Institute of Marine Research and ICES. Also, it was considered that the Integrated Taxonomic Information System (ITIS) codes are suitable to be used for the Baltic phytoplankton.

The distribution of a questionnaire on the conduct of primary production studies had been delayed since there were a number of developments after the 2002 SGQAB and SGQAE meetings that needed to be incorporated. The questionnaire was revised and then reviewed by the joint SGQAB/SGQAE group; responses of laboratories and individuals to the questionnaire have been received. The ACME suggests accelerating the finalization of work on this item.

SGQAB noted that HELCOM has postponed the establishment of a phytobenthos QA group. Phytobenthos monitoring is being conducted in Denmark, Estonia, and Sweden, while in Latvia and Finland first pilot studies have been made. Guidelines have already existed for several years and are followed in the countries implementing monitoring of phytobenthos. SGQAB decided to make further developments in the phytobenthos programme.

SGQAB reviewed the report of the HELCOM training course on phytoplankton and the outcome of the activities of the HELCOM Phytoplankton Expert Group. A checklist for Baltic phytoplankton species has been completed and is available on the Algaline website. The phytoplankton counting program has been adapted to the ICES Biological Data Reporting Format, and has been accepted by ICES. A new microscopy technology was presented and spring samples were intercalibrated during the annual meeting of the Phytoplankton Expert Group in

Germany. SGQAB supported the idea of endowing rights to allow OSPAR groups monitoring phytoplankton to use the HELCOM phytoplankton counting program.

SGQAB welcomed the new Zooplankton Expert Network, expecting active work of the network and continual renewal of a zooplankton manual starting immediately. SGQAB revised Part B of the HELCOM COMBINE Manual and relevant technical annexes of Part C of this manual. The ACME endorsed these changes to the COMBINE Manual for transmission to HELCOM.

In a joint session of SGQAB and SGQAC, the QA Guidelines for measuring primary production and chlorophyll *a* were discussed. It was decided that relevant chapters should be inserted into the HELCOM COMBINE Manual. A SGQAB/SGQAE session, in turn, decided that ICES should inform ISO and CEN about the current QA activities of SGQAB, SGQAE, and SGQAC.

The most recent developments by countries concerning ring tests and intercalibration exercises were also reported. A German representative presented the results of the First National Chlorophyll *a* Ring Test. The samples were analysed by three different methods: 1) the German Standard (DIN 38412-16); 2) the HELCOM COMBINE Manual (C-4) method; and 3) the laboratory-specific routine methods of the participants. The results obtained testified that all three methods are appropriate to determine the chlorophyll *a* content (standard deviations 12–31%). However, none of them is fit for the determination of phaeopigments (standard deviations 87–158%). A representative of the UK reported about a test developed for macrozoobenthos species recognition and based on a website. Representatives of Lithuania and Latvia reported about the HELCOM area phytoplankton intercalibration (all HELCOM member countries were represented) and intercalibration of phytoplankton and zooplankton sampling (participation by three eastern Baltic countries), respectively.

5.2 Quality Assurance of Biological Measurements in the Northeast Atlantic

Request

This work has previously been conducted in response to a request from the OSPAR Commission, which was not renewed for 2003. However, given the importance of work to promote the implementation and improvement of quality assurance of biological measurements and monitoring performed within the ICES area, ICES decided to continue this work nonetheless.

Source of the information presented

The 2003 report of the ICES Steering Group on Quality Assurance of Biological Measurements in the Northeast Atlantic (SGQAE) and ACME deliberations.

Scientific background

At its 2003 meeting, SGQAE prepared a review on thirteen items. This review included the outcome of the 2002 ICES Annual Science Conference, including the reports of five Working Groups whose activities are relevant to quality assurance of biological data. Representatives of several countries reported on their QA activities (ring tests, intercalibrations) to SGQAE. It appeared that several countries are increasing their commitment to quality assurance of biological studies, but the trend does not fully cover all the countries within the ICES and OSPAR areas. SGQAE also explored the current activities and links with other international agencies (e.g., ISO, CEN, EC, BEQUALM) having an interest in QA of biological measurements. A review of the progress on the implementation of the ICES Biological Community Database was prepared. SGQAE also examined screening methods for the analysis of biological data (phytoplankton, phytobenthos, zoobenthos) and for determining the acceptability of biological sampling and analytical practices. The QA/AQC implications for a large application of environmental indicators employing biological measures were analysed.

When reporting on joint sessions and activities, common among SGQAE/SGQAB/SGQAC, emphasis was given to the possibility of laboratories obtaining common accreditation for biological and chemical activities, as has already been developed in several ICES Member Countries.

The ACME noted that OSPAR had ceased its financial support to SGQAE activities in 2003, at a time when the Steering Group has revised its terms of reference and prepared a strategy for the future which includes better lines of communication with relevant bodies (OSPAR, EC), ensuring that the requests are efficiently dealt with. At present, several biological measurements are not yet covered by QA procedures, and several OSPAR countries still need to implement or improve the quality assurance of biological measurements which are in use to fulfill the corresponding OSPAR monitoring requirements (e.g., Ecological Quality Objectives, eutrophication status). There is a strong need to support the development of quality assurance at an international level within the OSPAR area. The activities of SGQAE, as they are expressed in their revised terms of reference, will be essential to implement QA of biological measurements, where and when necessary. To enhance the value of SGQAE work to OSPAR, the ACME agreed that a large portion of its work should relate to the implementation of relevant activities under the OSPAR Joint Assessment and Monitoring Programme.

Recommendations and advice

ICES recommends that financial support should again be provided by OSPAR to the activities of SGQAE in the coming years.

ICES recommends that Member Countries continue active implementation of QA procedures for biological measurements and monitoring.

5.3 Quality Assurance Procedures for Biological Effects Techniques

Request

This is part of continuing ICES work to improve the tools available for monitoring contaminants and their effects in the marine environment. This issue is of particular relevance to the OSPAR Commission with regard to the implementation of biological effects monitoring under the Coordinated Environmental Monitoring Programme.

Source of the information presented

The 2003 reports of the Working Group on Biological Effects of Contaminants (WGBEC), the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO), and ACME deliberations.

Summary

The ACME took note of the continuing developments in the BEQUALM (Biological Effects Quality Assurance in Monitoring) programme, which was initiated via WGBEC in 1998 and funded by the EU Standards, Measurement and Testing Programme (SMTP). Progress in BEQUALM has been reported annually at ACME meetings since 1999.

Following the developments within BEQUALM, OSPAR is considering changing the categories of the three Technical Annexes dealing with liver histopathology, liver nodules, and externally visible fish diseases from Category II to Category I.

The BEQUALM programme will continue as a self-funding scheme focusing initially on three areas, namely, biomarkers, whole organism measurements, and community structure.

The steering group of BEQUALM has agreed to meet with the QUASIMEME steering group on an annual basis to discuss work programmes and to ensure communication and agreement on areas of mutual interest. In this respect, QUASIMEME will take forward QA/QC programmes in 2003/2004 for bile metabolites, chlorophyll determination, and imposex.

Recommendations and advice

ICES recommends to Member Countries that laboratories involved in the monitoring of biological effects in the marine ecosystem should participate in relevant parts of the BEQUALM quality assurance scheme, including laboratories engaged in environmental monitoring programmes using externally visible fish diseases and/or liver histopathology. Progress should be reviewed at regular intervals and the programme amended accordingly.

As a result of the continued development of integrated chemical and biological effects monitoring, ICES recommends that there be close liaison between the steering groups of the relevant QA schemes (QUASIMEME and BEQUALM).

ICES recommends that, where appropriate, the BEQUALM steering group consider organizing workshops covering specific themes and, over the years, the performance of laboratories should be assessed with a view to determining the overall impact of the BEQUALM programme.

ICES recommends that OSPAR upgrade to Category I the OSPAR Technical Annexes covering liver histopathology, liver nodules, and externally visible fish disease as a result of QA schemes now being available.

Scientific background

The main aim of BEQUALM is to provide a quality assurance scheme for biological effects techniques that are used in monitoring programmes such as the OSPAR Coordinated Environmental Monitoring Programme (CEMP), one of the three OSPAR data collection programmes which provide information for the assessments required under the OSPAR Joint Assessment and Monitoring Programme (JAMP).

The OSPAR Strategy with Regard to Hazardous Substances requires an assessment of any unintended/unacceptable biological responses, or levels of such responses, being caused by exposure to hazardous substances.

OSPAR Monitoring Guidelines are assigned a category. Category I guidelines are those for which quality assurance procedures are in place. Category I guidelines may be used for monitoring and the data obtained are appropriate for Convention-wide assessments. Category II guidelines are those for which quality assurance procedures are not yet in place. Category II guidelines may be used for monitoring, although caution should be exercised when making comparisons of the data obtained between different Contracting Parties.

OSPAR CEMP-rated biological effects issues relate to PAHs, metals, and organotins. In this respect, relevant biological effects techniques include:

- cytochrome P4501A;
- DNA adducts;
- PAH metabolites;
- liver histopathology;

- externally visible fish disease;
- metallothionein;
- ALA-D;
- oxidative stress;
- imposex and intersex.

Many of these techniques, with the exception of TBT-specific biological effects measurement which is Category I, are rated Category II by OSPAR. However, considerable progress has been made in respect of the quality assurance for externally visible fish diseases and liver histopathology under Work Package 9 (liver histopathology, liver nodules and externally visible fish disease measurement) of BEQUALM, the final report for which was presented to the 2003 meeting of the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO).

The outcome of BEQUALM Work Package 9 is a set of agreed protocols encompassing every procedure from collection of fish samples, handling, external disease assessment and recording, sampling for organ histopathology, laboratory procedures, diagnostic criteria, and reporting. All of these are explained on a CD-ROM and in the associated workshop reports.

OSPAR is now proposing to change the categorization of the Technical Annexes covering liver histopathology, liver nodules, and externally visible fish diseases from Category II to Category I. The ACME supports this change.

The EU funding for the BEQUALM programme ended in August 2002 and the programme is now being taken forward as a self-funding scheme. It consists of a steering group, which oversees, develops, and implements annual QA/QC programmes in three areas: a) biomarkers; b) whole organism; and c) community analysis. The programmes for 2003/2004 and 2004/2005, and lead laboratories, are summarized in Table 5.3.1.

5.4 Quality Assurance of Chemical Measurements in the Baltic Sea

Request

Item 1 of the 2003 requests from the Helsinki Commission: to coordinate quality assurance activities on biological and chemical measurements in the Baltic Sea 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 2003 reports of the ICES/HELCOM Steering Group on Quality Assurance of Chemical Measurements in the Baltic Sea (SGQAC), the Marine Chemistry Working Group (MCWG), and ACME deliberations

Table 5.3.1. Summary of the now self-funding BEQUALM programme which commenced on 1 May 2003 (www.bequalm.org). Results of the 2003/2004 programme will be presented at the 2004 meeting of WGBEC.

Area	**Lead Laboratory**	**Programme**	
		2003/2004	**2004/2005**
Biomarkers	NIVA	EROD/CYP1A1 AChE VTG in cod/rainbow trout Protein	Metallothionein ALA-D DNA adducts "Other biomarkers"
Whole organism	CEFAS	Externally visible fish diseases and liver histopathology *Tisbe battagliai* Oyster embryo Turbot juvenile acute *Corophium* whole sediment *Arenicola* whole sediment TIE (depending on participation)	*Daphnia* acute and chronic *Acartia* acute *Skeletonema* 72-hr growth
Community structure	Scottish EPA (NMBAQC)	Benthic community analysis	Phytoplankton assemblage

Abbreviations: EROD: ethoxyresorufin-*O*-deethylase; AChE: acetylcholinesterase; VTG: vitellogenin; ALA-D: δ-aminolevulinic acid dehydratase; TIE: Toxicity Identification and Evaluation.

Scientific background

The ACME reviewed the work of the ICES/HELCOM Steering Group on Quality Assurance of Chemical Measurements in the Baltic Sea (SGQAC), noting further progress by SGQAC in the development of QA provisions in relation to the Guidelines for the HELCOM Cooperative Monitoring in the Baltic Marine Environment (COMBINE) Programme.

A number of Technical Notes have been completed by SGQAC and are proposed for inclusion in the COMBINE Guidelines Manual, as listed below. The ACME accepted these Technical Notes for transmission to HELCOM.

A Technical Note on the Determination of Measurement Uncertainty was reviewed and commented on by the Marine Chemistry Working Group (MCWG). MCWG felt that it was a very positive sign that SGQAC recognizes the importance of taking into account uncertainty when evaluating monitoring data. However, the ACME considered the document to be too theoretical, and recommends that it be elaborated by providing practical examples that could be used in the laboratory. It was also considered that the inclusion of definitions in the document would be beneficial to the user. It is anticipated that the Technical Note will be completed in 2004.

Recommendations and advice

ICES recommends that the following Technical Notes, as completed by SGQAC, be included in the HELCOM COMBINE Manual:

1) Technical Note on the Determination of Heavy Metals in Marine Sediments;
2) Technical Note on the Determination of Polycyclic Aromatic Hydrocarbons (PAHs) in Sediments;
3) Technical Note on the Determination of Polycyclic Aromatic Hydrocarbons in Biota;
4) Technical Note on the Determination of Chlorobiphenyls in Sediments; and
5) Technical Note on the Validation of an Established Analytical Method for Chemistry.

5.5 Developments within QUASIMEME

Request

On account of long-standing ICES involvement in quality assurance matters, this is an ACME initiative to follow the developments in this QA project.

Source of the information presented

The 2003 report of the Marine Chemistry Working Group (MCWG) and QUASIMEME reports.

Summary

QUASIMEME (Quality Assurance of Information in Marine Environmental Monitoring) has completed ten years (1992–2003) of support for the external quality assurance for marine institutes for chemical measurements. Participation in the QUASIMEME Laboratory Performance Study Scheme is open to all institutes, worldwide, that conduct chemical measurements in sea water, sediments, or biological materials. The following is a summary of ongoing and proposed QUASIMEME activities.

The QUASIMEME routine laboratory performance studies continue on a number of substances in sea water, sediments, and biota.

The QUASIMEME development exercises for 2003–2004 have a new approach:

- Organotin compounds will have a fresh focus. Regular exercises will also include investigation of the different steps in the analysis to identify the main sources of error. This will be done over a period of two to three years. There will also be sufficient matrices included for institutes to offer routine external QA to underpin the national and international marine monitoring programmes.
- With regard to shellfish toxins, QUASIMEME is commencing a development programme to underpin the measurement of shellfish toxins. In the first instance, the programme will focus on the measurement of amnesic shellfish poisoning (ASP) toxins and later extend to diarrhetic shellfish poisoning (DSP) toxins and paralytic shellfish poisoning (PSP) toxins. This is being done in conjunction with the EU Reference Laboratory for Marine Biotoxins in Vigo, Spain, the Marine Institute, Galway, Ireland, and the FRS Marine Laboratory, Aberdeen (UK National Reference Laboratory for Biotoxins).
- Brominated flame retardants will continue to be measured in biological tissues and measurements will be extended to include sediments.

The QUASIMEME Workshops planned for 2003–2004 are:

1) Organochlorine Pesticide Residues in biota and water (proposed);

2) Particulates and Filtration – Metals in water (proposed);

3) Shellfish Toxins, Galway, Ireland, June 2004;

4) Imposex and Intersex in Marine Gastropods, The Netherlands, September/October 2003;

5) Proposed joint QUASIMEME/RIKZ/OSPAR workshop.

Information on QUASIMEME is on the website http://www.frs-scotland.gov.uk.

Recommendations and advice

ICES recommends that monitoring laboratories in Member Countries adhere to established QA protocols and report QA information with environmental data submitted to the ICES Marine Data Centre.

5.6 Standardized Presentation of the Long-term Performance of a Laboratory

Request

This is part of continuing ICES work to provide information on methods to aid in the assessment of environmental monitoring data, and is particularly relevant to temporal trend assessments.

Source of the information presented

The 2003 report of the Marine Chemistry Working Group (MCWG) and ACME deliberations.

Summary

Of the twenty groups of determinands studied in the QUASIMEME Project in 1996–2002, laboratory performance on most of these showed an overall improvement in the between-laboratory agreement. In some cases, the improvement was modest and still requires further attention. However, the value of long-term participation of chemical laboratories in an external QA programme has clearly been demonstrated.

Recommendations and advice

ICES encourages the participation of chemical laboratories in Member Countries in QA exercises such as those coordinated by QUASIMEME.

Scientific background

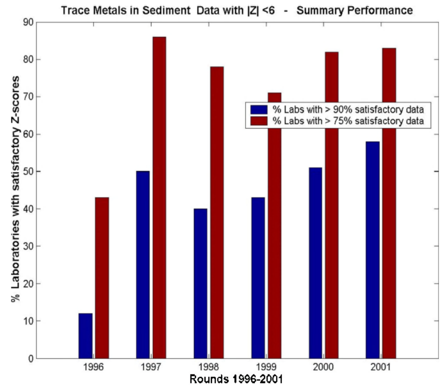
A paper was presented at the QUASIMEME Conference held in Barcelona in 2002 that provided an overview of progress within the QUASIMEME project from 1996–2002 (Asmund *et al.*, in press). A series of Quality Performance Indicators were used to indicate year-by-year progress.

On a laboratory basis, two categories are defined within any specified group, e.g., nutrients in sea water, or trace metals in biota:

1) % of laboratories that have > 90 % satisfactory performance (| Z | < 2) for a specified group of determinands;

2) % of laboratories that have > 75 % satisfactory performance (| Z | < 2) for a specified group of determinands.

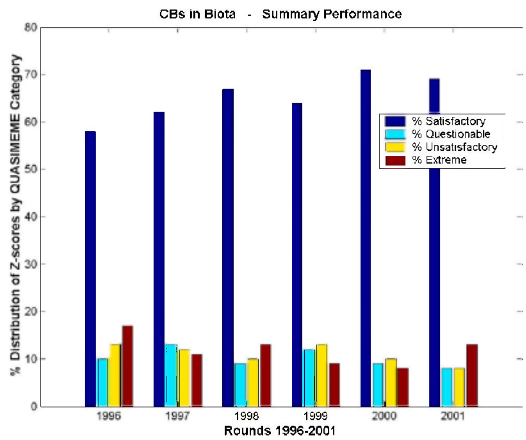
An example is given below for trace metals in sediment:



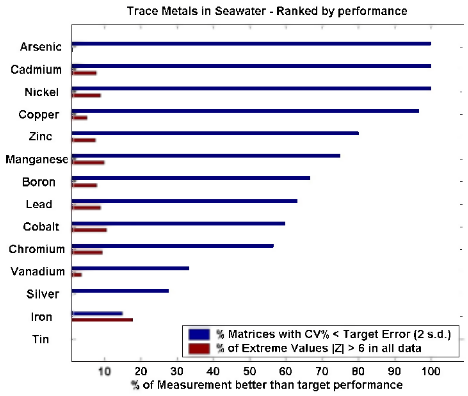
A second classification used the standard approach of ISO 43 in grouping laboratory performance:

- % of laboratories with satisfactory performance (| Z | < 2);
- % of laboratories with questionable performance (2 < | Z |< 3);
- % of laboratories with unsatisfactory performance (3 < | Z | < 6);
- % of laboratories with extreme values (| Z | > 6).

An example is given below for CBs in biota:



On a determinand basis, a comparison has been made on the actual between-laboratory performance and the target performance for each determinand group. An example is given below for trace metals in sea water:



The long-term precision and bias can be determined on the basis of a re-scaled sum of Z-scores over a prescribed time period, to map the long-term performance. This approach is now part of the ISO 43 guidelines for laboratory performance.

Of the twenty groups of determinands studied in the QUASIMEME Project from 1996–2002, most show an overall improvement in the between-laboratory agreement. In some cases, the improvement was modest and still requires further attention. However, the value of long-term participation in an external QA programme has clearly been demonstrated.

Each of these approaches has been completed by the QUASIMEME project for the period 1996–2002 and is currently being compiled for participants in the Laboratory Performance Scheme or the Conference on CD-ROM. This is available to others on request. A detailed overview is also being prepared for publication.

Reference

Asmund, G., Scurfield, J., and Wells, D. (In press). Treatment of laboratory performance study data. Proceedings of the QUASIMEME Conference Measurement in the Marine Environment: Challenges and Achievements, Barcelona, 23–26 October 2002.

Total Groups = 20	**Performance by laboratory**	**Overall performance by determinand group**
Improved	11	13
Consistently high	5	3
Poor, erratic or declined	4	4

6 ENVIRONMENTAL ASSESSMENT: METHODOLOGIES AND RESULTS

6.1 National Sediment Quality Criteria Values and their Derivation

Request

This is part of ongoing ICES work to provide scientific advice on marine environmental quality objectives to ICES Member Countries, OSPAR, and HELCOM. It is relevant to all authorities that are developing standards for use in regulatory and management procedures regarding the marine environment.

Source of the information presented

The 2003 report of the Working Group on Marine Sediments in Relation to Pollution (WGMS) and ACME deliberations.

Scientific background

In 2002, the ACME reviewed the first material compiled by WGMS for an inventory of Sediment Quality Criteria (SQC) values and standards for dredged material disposal in ICES Member Countries. The purpose is that the inventory may allow for a comparison of values and approaches applied, and possibly contribute to SQC harmonization in the ICES area.

An updated and revised document was considered and finalized by WGMS at its 2003 meeting. WGMS noted that the information compiled shows that some countries (e.g., Canada, The Netherlands, Norway, and Sweden) have developed Environmental Quality Standards (EQS) for sediments (law-enforceable regulations), and that several countries are currently producing or revising approaches towards EQS and Sediment Quality Guidelines (SQGs). The information on dredged material shows some consistency among the concentrations set for target values by the different countries, but the ranges in limit values are much wider. WGMS suggested that these large differences may reflect local or regional conditions.

The introduction to the document gives a summary review of several different approaches for setting Sediment Quality Guidelines, pointing out their main advantages and disadvantages. A few references are also included. The ecotoxicological approaches Threshold Effects Levels/Probable Effects Levels (TEL/PEL) and effects range low/effects range medium (ERL/ERM) are considered the most promising options, with the latter simpler to operate and having received more validation. The Sediment Quality Triad is also mentioned as promising, but it is pointed out that the methodology to integrate data to give a reliable weigh-of-evidence is not fully developed in this case, and the approach is probably too data-hungry to form the basis for operational sediment assessment for more than a handful of substances and locations.

The ACME endorsed the view, based upon Burton (2002), that Sediment Quality Guidelines should be used as part of a holistic assessment in a screening manner or in a “weight-of-evidence approach”, in which multiple components are assessed (e.g., habitat, hydrodynamics, resident biota, toxicity, and physico-chemistry, including SQGs) by using integrated approaches.

The ACME noted that the inventories, although more complete than last year, seem to have similar types of problems and gaps as those pointed out by ACME in 2002 (ICES, 2003). It was noted that, at present, the Dredged Material section has a much larger number of contributions by ICES Member Countries, compared to the Sediment section. For the latter, a detailed presentation on approaches, levels, and status of sediment SQGs was made by only a few countries, and several report on work under way.

Comments made by ACME in the past on the advantages and drawbacks of the different approaches to derive SQGs, and on ways forward (ICES, 2001, 2003), are considered relevant and up-to-date for the present scientific advice on this issue. In particular, weaknesses identified were, e.g., that there are few data available on whole sediment tests, and only for a limited range of robust organisms; there are major difficulties in estimating bioavailability for chemical and physical properties; and, when field data are used to derive criteria, there are problems in quantifying the contribution from each contaminant, and distinguishing this from natural factors. The lack of more sensitive biological endpoints than immobilization or death in the sediment bioassays was noted.

The knowledge on biomarker-type endpoints for organisms used in the tests is rapidly growing. For sediments, this is already being used particularly in projects and case studies. More information is found in Section 4.1.3, above. Validation of methods for biological testing is also in progress, and is considered a major point for further work.

The ACME considered that the document “Inventory of Sediment Quality Criteria in ICES Member Countries” contains useful information for both ICES Member Countries and the regulatory Commissions; thus, it is attached as Annex 3.

Recommendations and advice

ICES recommends that the document “Inventory of Sediment Quality Criteria in ICES Member Countries”, as contained in Annex 3, be considered by OSPAR and HELCOM for use in their assessments of sediment quality and dredged material.

References

Burton, G.A., Jr. 2002. Sediment quality criteria in use around the world. Limnology, 3: 65–75.

ICES. 2001. Report of the ICES Advisory Committee on the Marine Environment, 2001. ICES Cooperative Research Report, 248: 50–51.

ICES. 2003. Report of the ICES Advisory Committee on the Marine Environment, 2002. ICES Cooperative Research Report, 256: 13–14.

6.2 Procedures for Conducting Temporal Trend Assessments of Data on Contaminants in Biota and Sediments

Request

This is part of continuing ICES work on methods for temporal trend monitoring of contaminants in biota and sediments, and the implications for marine environmental programmes in the ICES area.

Source of the information presented

The 2003 report of the Working Group on Statistical Aspects of Environmental Monitoring (WGSAEM) and ACME deliberations.

6.2.1 Development of trend assessment for test against background

Summary

WGSAEM discussed how to conduct an appropriate test of whether the annual mean contaminant concentration is at or below some reference value. In particular, WGSAEM considered a method which provides a test that reduces the type II-error when the true mean is above the background reference concentration (BRC). The theory behind the test is quite general, but it can easily be formulated in terms of the assessment tools described below (OSPAR, 2002).

The problem is that, by definition, the mean cannot fall below background. This implies that we have to use a classical test, i.e., of whether the observed mean is significantly greater than background and hence rejecting the null hypothesis that the mean is at background. However, with poor monitoring precision, this test may allow considerable contamination before the null hypothesis is rejected. A solution more consistent with the precautionary approach is to define a *near-background* concentration. The test then assumes a null hypothesis that the mean is above the *near-background* concentration, which is rejected if the mean is significantly below this value.

In conclusion, the ACME agreed the following:

- An appropriate value for the *near-background* contaminant concentration needs to be identified for each specific contaminant in each matrix monitored;
- Consistency between the values chosen for environmental objectives and the power of monitoring programmes is important.

Scientific background

Ideally, the *near-background* concentration of a contaminant would be defined on an ecotoxicological basis, for example, as the no-effect concentration, or as a concentration consistent with an acceptable level of contaminant effect. In practice, a more pragmatic approach may be necessary if there is no basis for an ecotoxicological choice. One method (described in Annex 4) is to link the precision of the monitoring programme to a corresponding value of the *near-background* concentration, solving for the value that would have a 90% chance of being rejected when the true mean concentration equals the BRC. The method was demonstrated using an example of mercury concentrations reported in plaice from the Irish Sea. Within the OSPAR protocol, the test simply focuses on the upper 95% confidence limit of the predicted mean concentration in the final year of the time series.

This issue extends to situations where there are several reference points dividing the contaminant range into a sequence of levels, e.g., the *High*, *Good*, *Moderate*, *Poor*, and *Bad* classifications for ecological status employed in the EU Water Framework Directive for biological, physico-chemical, and hydromorphological elements.

6.2.2 Standard assessment protocol for trends

Summary

WGSAEM considered a proposal for a standard assessment protocol for trends in contaminant concentrations in biota that had been presented at the meeting of the OSPAR Working Group on Monitoring (MON) in December 2002 (OSPAR, 2002). This protocol was a refined version of the method used in the 1998 OSPAR temporal trend assessment, and had subsequently been presented to the ICES Secretariat to consider the development of appropriate software to automate the analysis. This would replace the SAS software used in the 1998 assessment, and be enhanced to produce automated tables of the various *ad hoc* summaries that had been produced in the past. WGSAEM was invited to comment on aspects of this protocol that could be further improved or developed, and to share their experience of data-induced problems that had occurred in other large-scale assessments and suggest how these could be avoided. WGSAEM also discussed the protocol that was used in the report of the ICES/AMAP Study Group for the Assessment of AMAP POPs and Heavy Metals Data (ICES, 2002).

The ACME concluded that:

a) regional trend plots should be accompanied by a map showing the locations of monitoring sites;

b) the various graphs, summary statistics, and analyses of variance tables for a contaminant within a region should be collected together;

c) statistics such as the detectable trend (after ten years) and the slope of the recent trend (over a seven-year span) should be aligned to a common duration;

d) where possible, a contrast between the estimated mean in the final year and the mean in some reference year would enhance the results and should be included amongst the summary indicators;

e) additional covariates such as liver fat weight should be considered;

f) alternative estimators of the residual variance that are independent of the choice of span in the fitted smoother should be considered;

g) some assessment of outliers could be included; partly this might improve the estimate of trend and the power of detection, and partly it might provide additional information on the frequency of isolated incidents such as spillages.

Scientific background

The following issues were explored:

1) the effect of outliers on trend detection;

2) the effect on trend detection of allocating different values for undetected observations.

Outliers

The issue of outlier detection and interpretation is of general interest. There was a general consensus that, where outliers are evident, it may be possible and useful to partition the trend into: a) a smooth component, and b) isolated "incidents" where outliers are large and on the high side.

The time series contained various numbers of observations per year, but have been analysed in a consistent way:

- The annual geometric mean concentration is computed, with values reported as below the limit of detection being allocated a value estimated from the data.
- A linear trend (on a log-scale) is fitted to the annual geometric means and tested assuming that year-to-year variation is homoscedastic and Normally distributed.
- Outliers are tested using the method of Grubbs (Grubbs, 1969) where deleted-residuals are tested and the largest significant deleted-residual is deleted, a new set of deleted-residuals is computed with this value omitted, and so on until there are no further significant deleted-residuals. (A deleted-residual is the difference between an observation and a linear fit to the data having omitted the observation.)

The effect of outliers on trend detection was demonstrated for trends in lead in herring livers using two time series collected by Sweden from the Baltic Sea and the Kattegat (Figure 6.2.2.1). For the Harufjärden time series, high leverage outliers resulted in an overall trend of 5% compared with an outlier-deleted trend of 7%. Removing the outliers also increased the short-term trend. For the Utlängan time series, the outliers had a smaller effect on the estimated trends; however, the outliers had the effect of increasing the residual variance, making the small trend not statistically significant.

This demonstrates that outliers have potentially two effects: distorting the trend and/or masking its significance. By excluding possible outliers, the sensitivity of detecting a trend in a ten-year time series with 80% power improved from an annual change of 9.7% to 5.5% and 13% to 6.2% for Harufjärden and Utlängan, respectively. Alternatively, the number of years required to detect a 5% annual change with a power of 80% decreased from 15 years to 11 years and from 18 years to 12 years, respectively.

There are several technical details that need to be resolved though, before some single procedure can be recommended for routine, automated use within an assessment.

A major question is whether outliers should be deleted or simply identified. If statistically significant outliers are routinely deleted, the residual variance will tend to be underestimated, increasing the type I error rate of trend detection. However, depending on the pattern of trend, the type II error rate could also be affected. Both the type I and type II error rates will also tend to be distorted if outliers are not deleted. Further work on the interaction between patterns of trend and the type I and II error rates is required.

Secondly, the presentation of results will need to be modified so that information about outliers is presented in a way that meets the requirements of the monitoring agency (e.g., OSPAR). For example, they may wish to have an assessment of the trend fitted to the data as reported. Alternatively, they may wish to present an outlier-deleted trend and identify any extreme high values as possible contaminant incidents. This has to be resolved in negotiations with the agencies.

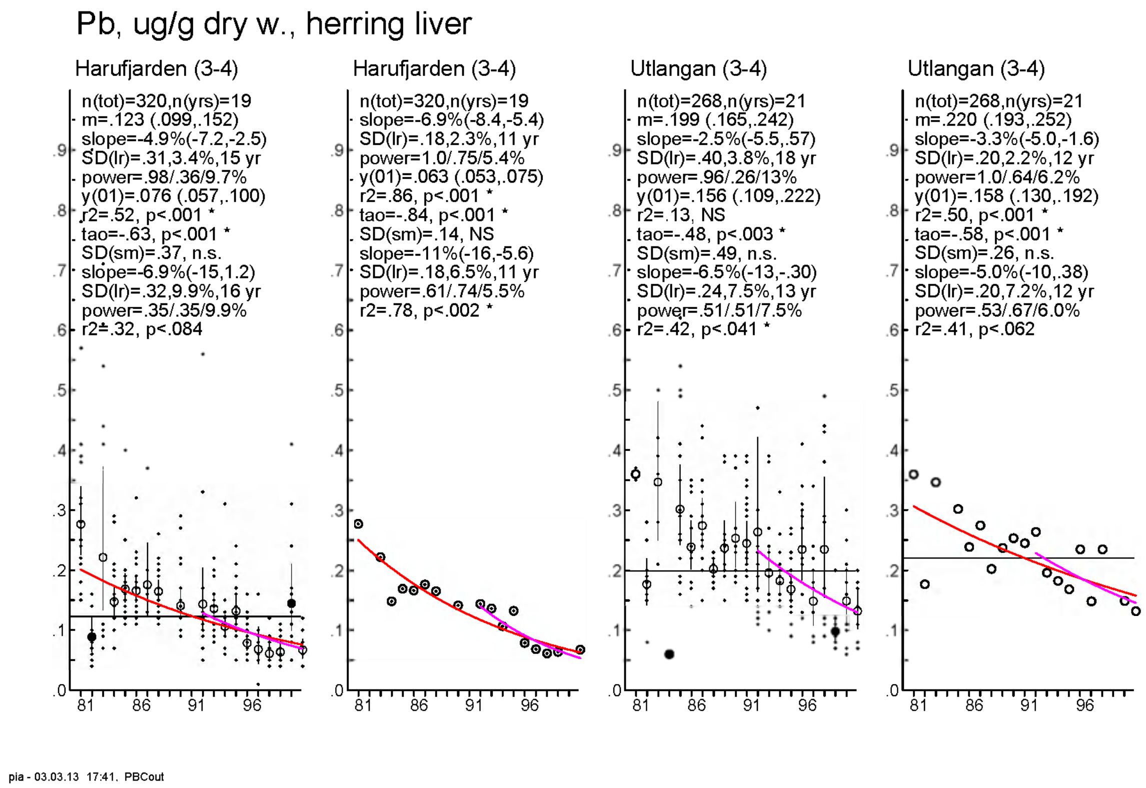


Figure 6.2.2.1. The effect of outliers on trend detection for two time series of lead in herring livers. For each time series, there is a pair of figures. In the left-hand figure, the complete data set is shown together with both the overall and final ten-year trends. The geometric means detected as outliers are shown as solid circles. The right-hand figure shows the geometric means with the outliers removed and the corresponding long- and short-term trends.

The effect on trend detection of different ways of treating results reported as below the detection limit

There are many statistical techniques for treating censored data when the true detection limit is known. The problem can sometimes be avoided by using a robust statistic such as the median, which is unaffected by small numbers of observations below the detection limit. An alternative is to replace the observations below the detection limit with a sensible value. For example, a common method is to allocate half the value of the detection limit. In these cases, the estimated annual mean concentration will depend on both the detection limit and the value allocated to non-detected results in the data set. In general, the estimate of the true mean value will be biased.

Concerning the problems induced by results reported as less than some limit of detection, there are several difficulties including:

- There appears to be no universal, consistently used criterion for testing detection or labelling the results. Hence, adopting a unified statistical treatment is difficult. Provided that the frequency of “less thans” within a year is low, using the median avoids the issue.
- If the analytical method is too coarse for the typical concentrations of the contaminant being monitored, even the median begins to fail.
- The value reported as the limit of detection often appears optimistically low relative to the distribution of detected results. These appear as extreme, low outliers, especially when results are expressed on a log-scale.

Two alternative approaches were considered. The first simply replaced observations below the limit of detection with half the smallest detected value. It was considered that this value is superior to half of the limit of detection, which often results in values that appear very detached from the main body of data, especially when expressed on a log-scale.

The second method estimated each unknown concentration using the empirical expected order statistic. This method fits a log-linear regression of the ranked detected concentrations on rank, and then uses this relationship to predict the value of those concentrations reported as below the limit of detection.

The methods were applied to the three data sets for CB52 collected by Sweden from the Baltic Sea and the Kattegat (Figures 6.2.2.2 and 6.2.2.3). For CB52, the

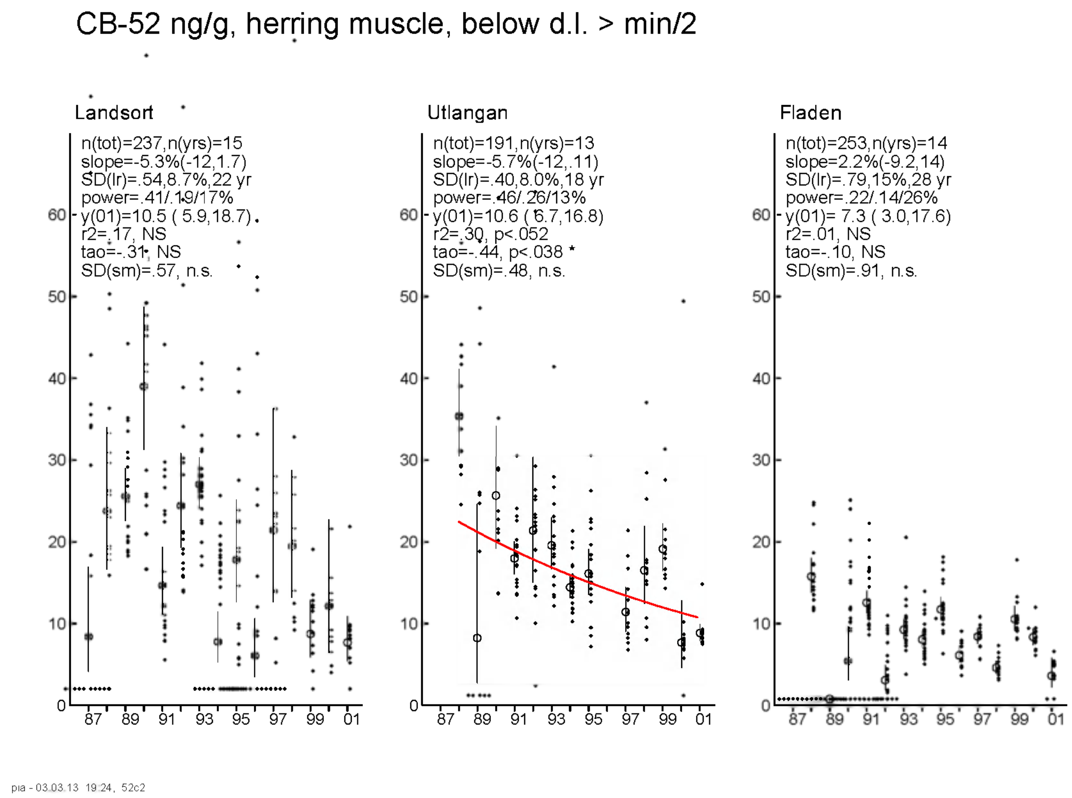


Figure 6.2.2.2. Effect of replacing observations below the limit of detection with half the minimum detected value.

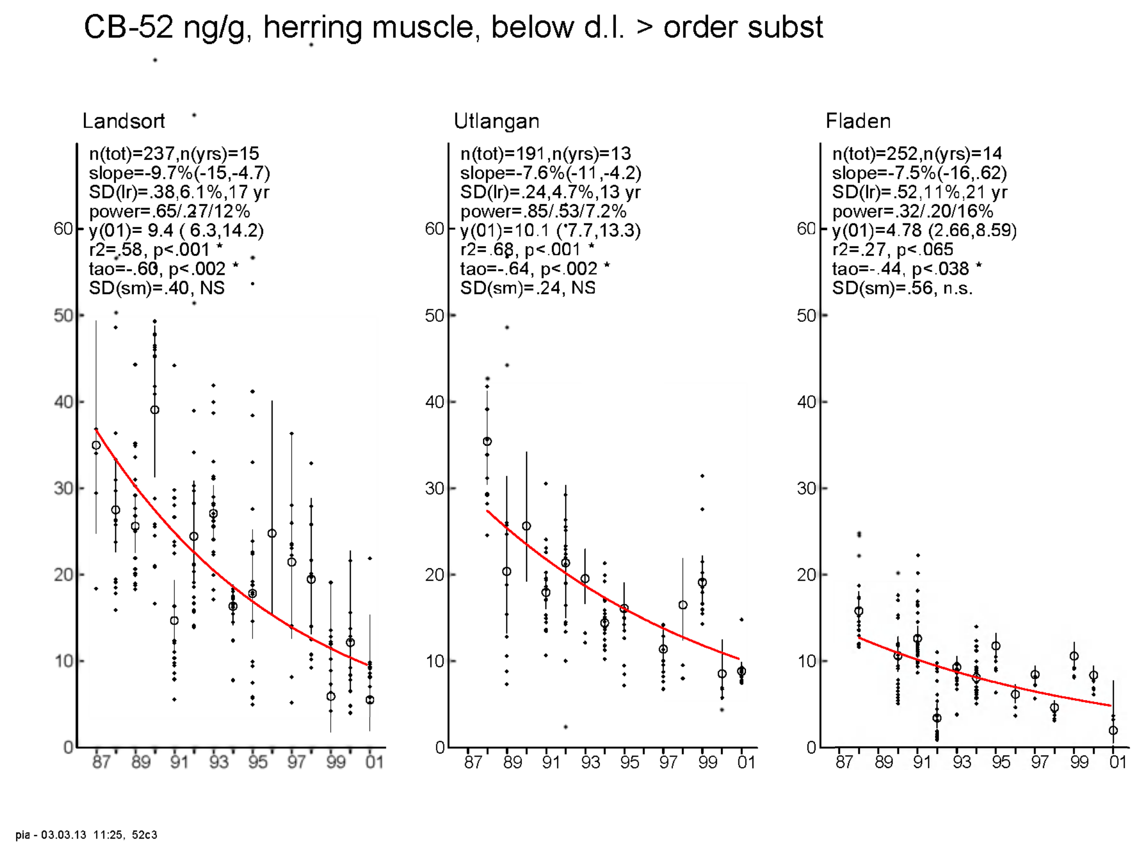


Figure 6.2.2.3. Effect of replacing observations below the limit of detection with estimates based on the empirical expected order statistic.

measured concentrations are close to the limits of detection, and the effects of different strategies are pronounced. With the regression-based method (Figure 6.2.2.3), there are significant log-linear trends at the 5% level for sites 1 and 2, and a marginally significant trend at the 10% level at site 3. Using the (minimum detected)/2 method (Figure 6.2.2.2), only the trend at site 2 is marginally significant. This lack of significance is probably due to the high frequency of non-detected values in the early years, which resulted in much lower annual means in these years.

The different methods have resulted in both different slopes and different residual variances. By applying the regression method, the sensitivity of detecting a trend in a ten-year time series with 80% power improves from an annual change of 17%, 13%, and 26% to 12%, 7.2%, and 16% for the three time series, respectively. Equivalently, the number of years required to detect a 5% annual change with a power of 80% decreased from 22, 18, and 28 years to 17, 13, and 21 years, respectively.

It was demonstrated that different methods of dealing with "less-than" values will have different effects on the estimated trends and their significance. WGSAEM concluded that both of the methods used were superior to (limit of detection)/2, in the sense that using (limit of detection)/2 would have produced values that were more detached from the body of the data. The regression method appeared to work well, but further evaluation is necessary.

Finally, it was noted that for these Swedish data, undetected values had simply been reported as undetected, and there was no choice but to estimate the concentration from the data or to allocate some nominal value. It would have been interesting to repeat this analysis using the actual number generated by the instrument and converted into concentration units. This procedure might provide an unbiased estimate of the annual mean concentration, although data produced this way might make chemists feel uncomfortable. This approach may also lead to a smaller standard error on the estimated annual mean concentration, since it avoids the extra variability introduced by dealing with censored data.

References

Grubbs, F.E. 1969. Procedures for detecting outlying observations in samples. Technometrics, 11: 1–21.

ICES. 2002. Report of the ICES/AMAP Study Group for the Assessment of AMAP POPs and Heavy Metals Data. ICES CM 2002/ACME:01.

OSPAR. 2002. Tools for routine assessments of trends in OSPAR CEMP data. Paper at the 2002 meeting of the Working Group on Monitoring. MON 02/5/2-E. (Available at http://www.ospar.org).

6.3 Trend Analysis of Monthly and Quarterly Data on Inputs of Nutrients and Contaminants to the Marine Environment

Request

This is part of continuing ICES work on methods for the temporal trend assessment of monitoring data on inputs of contaminants and nutrients via rivers or the atmosphere to the marine environment. It is particularly relevant for OSPAR and HELCOM programmes relevant to inputs and loads of contaminants and nutrients.

Source of the information presented

The 2003 report of the Working Group on Statistical Aspects of Environmental Monitoring (WGSAEM) and ACME deliberations.

Summary

WGSAEM considered three topics related to the analysis of monthly and quarterly input data and/or to the analysis of nutrient data:

1) Change-points reflect points in the time series where there is a break in the slope of the trend line. Change-points may appear, e.g., when reduction measures are imposed or when new sources of contaminants arise. A paper was discussed covering three different settings for the detection of change-points: a linear regression model, a regression model with seasonality, and a local regression approach (see Annex 5).
2) A document for Guidance on Input Trend Assessment and the Adjustment of Loads that had emerged from the OSPAR Workshop on Inputs, The Hague, 12–14 June 2002, was discussed. WGSAEM welcomed the guidance paper as an overview of potential methods to be applied in the trend assessment of inputs.
3) IMPACT, a recently finalized project within the EU research programme for Information Society Technologies, was discussed. The main objective of IMPACT was to develop new analytical tools that can facilitate the extraction of anthropogenic signals from environmental data influenced by meteorological conditions and other sources of natural variation. The project output comprised:
   - new statistical procedures for trend detection;
   - a theoretical framework for the incorporation of process-based deterministic models into statistical tools;
   - user-tested software;
   - short courses.

Conclusions

The ACME agreed that the methods and documents presented seem to imply promising tools to facilitate the evaluation of trend analysis. The next stage should be to see how the suggested methods perform in practice during full assessments of various input data, in particular, how they perform in relation to data quality, assessment software, and the presentation and interpretation of results.

Scientific background

6.3.1 Change-point models

Change-points reflect points in the time series where there is a break in the slope of the trend line. Change-points may appear, e.g., when reduction measures are imposed or when new sources of contaminants arise. In Annex 5, three different settings are considered for the detection of change-points: 1) a linear regression model, 2) a regression model with seasonality, and 3) a local regression approach.

The linear regression procedure provides a change-point and two regression lines with different slopes for fitting the data. Furthermore, a test is provided to decide whether such a fit is significantly better than an ordinary regression line. Such a situation is presented in Figure 6.3.1.1.

The linear regression procedure is suited for annual data, but it should not be applied to monthly or quarterly data (unless there is no seasonality). For monthly or quarterly data, the linear regression procedure is extended to include a seasonal term. An example is presented in Figure 6.3.1.2.

A linear trend is often not sufficient to model a very long time series. In this case, a LOESS-based method can be applied which allows an assessment of whether there is a significant break in the trend in addition to smooth fluctuations in the trend line. This is illustrated in Figure 6.3.1.3.

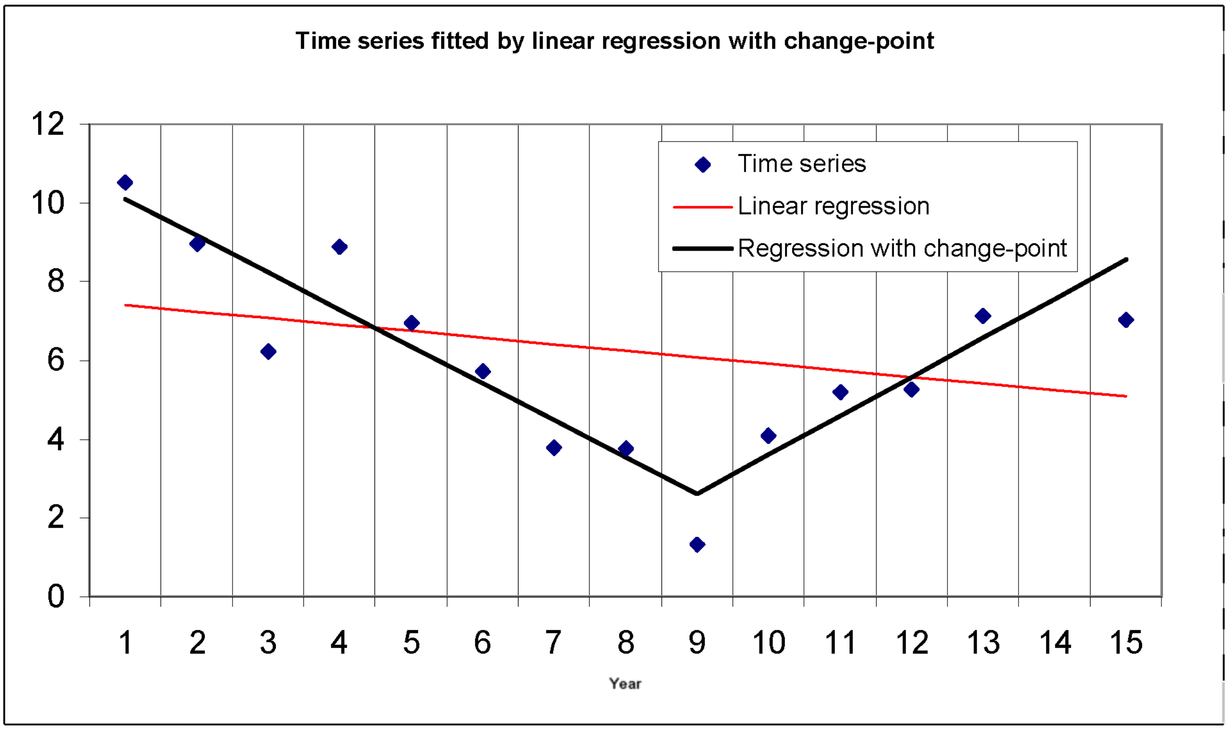


Figure 6.3.1.1. Illustration of linear regression change-point model.

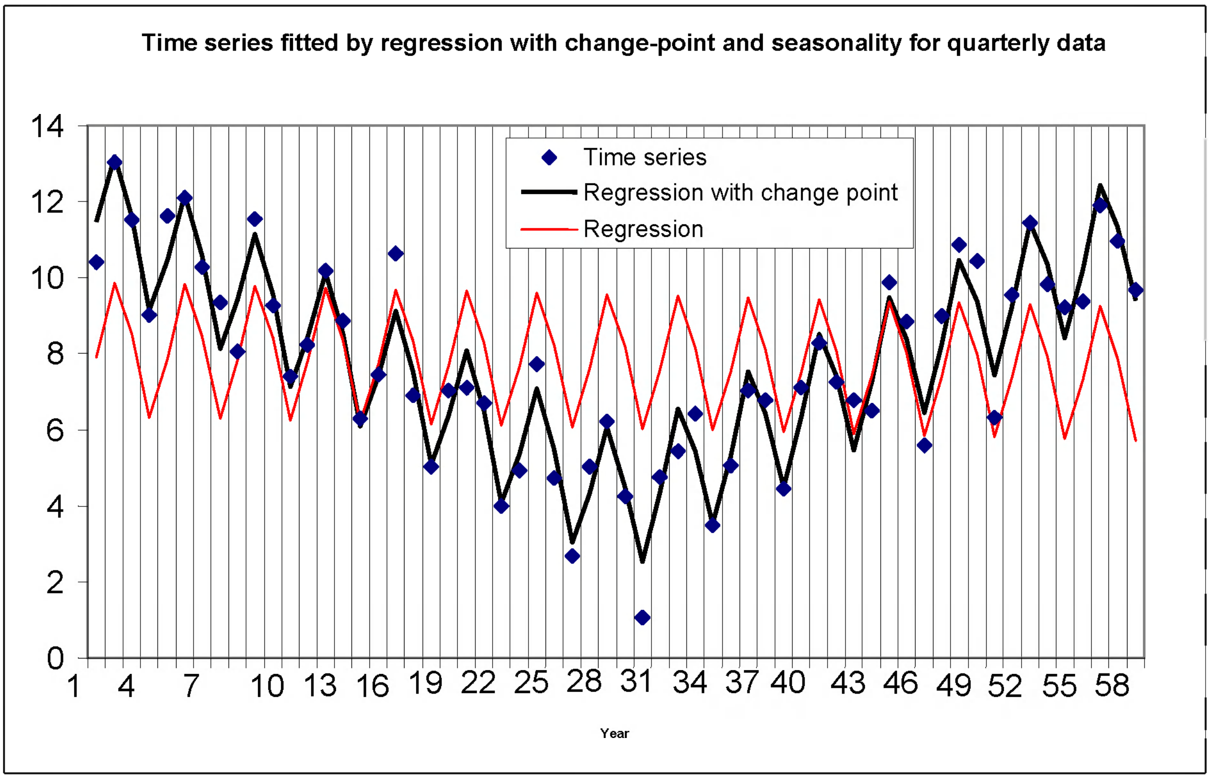


Figure 6.3.1.2. Illustration of linear regression with seasonality change-point model.

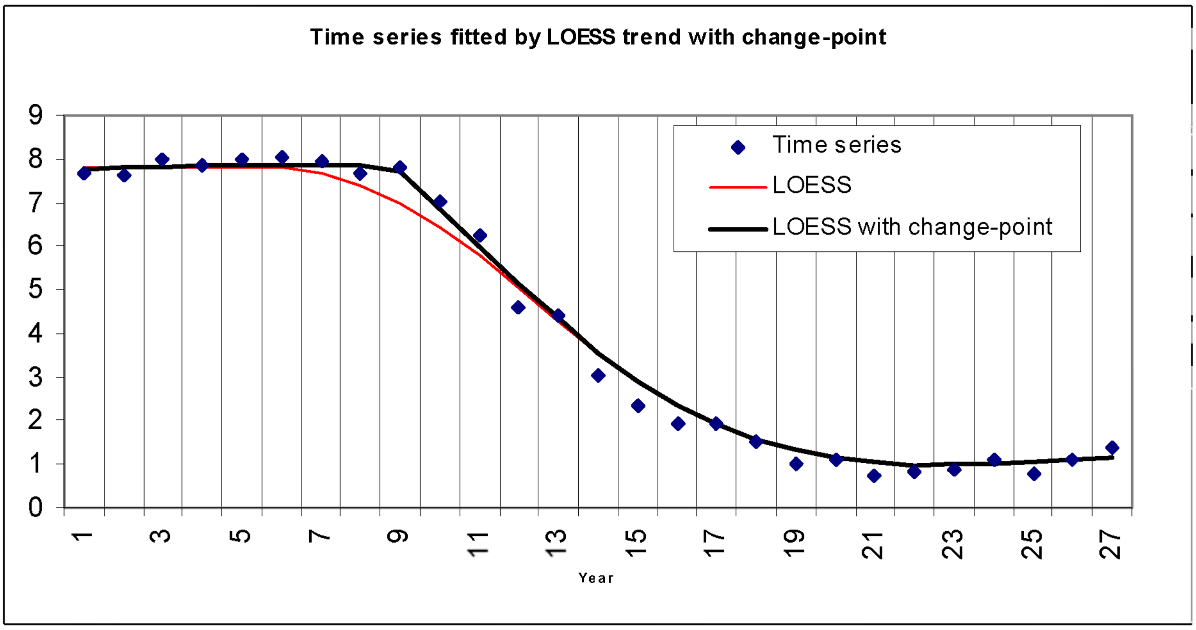


Figure 6.3.1.3. Illustration of LOESS-based change-point model, with a change-point in year 9.

The proposed procedure uses a significance test based on the F statistic with, e.g., 2 and n–4 degrees of freedom in the linear setting (where n is the number of years). These degrees of freedom arise because the change-point is considered an additional parameter that must be estimated in the linear regression model. The method provides reasonable results when the time series comprises ten to fifty years. With longer time series, the significance level for detecting a change-point is more than 7% when the nominal significance level is 5%.

In the review of these methods, it was noted that there are several tests for a change-point based on the likelihood ratio-statistic for which the theoretical distribution can be derived (e.g., Jaruskova, 1996; Gombay and Horvath, 1996). These authors consider the test statistic $Z_n^{1/2}$, defined as the square root of the log likelihood ratio of the model with and without a change-point. The classical chi-squared distribution does not appear to be adequate for deriving critical values for this test statistic. Its distribution has to be calculated by means of an asymptotic distribution (which is not identical to the chi-squared distribution) or by simulation.

The test statistic $Z_n^{1/2}$ is similar to the F statistic for the linear regression change-point model used in Annex 5, but it may also be used under non-Normal distributional assumptions and when there is autocorrelation. An advantage of the F statistic, however, is that no asymptotic calculations or simulation runs are required to provide critical values and it can be extended quite simply to non-linear settings.

Finally, it was also noted that, especially with fairly long time series, the presence of serial correlation will make all the tests discussed above too liberal, with substantial increases in the true significance level when there is autocorrelation of more than 0.5.

References

Gombay, E., and Horvath, L. 1996. On the rate of approximations for Maximum Likelihood Tests in change-point models. Journal of Multivariate Analysis, 56: 120–152.

Jaruskova, D. 1996. Change-point detection in meteorological measurement. Monthly Weather Review, 7: 1535–1543.

6.3.2 Guidance on input trend assessment and the adjustment of loads

A document entitled "Guidance on Input Trend Assessment and the Adjustment of Loads" from the OSPAR Workshop on Inputs, The Hague, 12–14 June 2002, was reviewed by WGSAEM. There was considerable discussion on the methods proposed.

WGSAEM recognized several open issues in the document, for example:

- How should the effect of measurements less than the limit of detection (LOD) be assessed? Should the ratio between the lower limit and the upper limit of the mean value be calculated on an annual basis or over the whole time series? And what happens when the ratio is below 0.6?
- What should be done when the LOD is not constant over the whole time series; e.g., should all data be discarded when the LOD is higher than an acceptable limit?
- How should one choose between the competing methods for adjustment with monthly or quarterly data? For example, there are six competing methods when sampling, concentration, and corresponding runoff data with long-term monthly mean runoff data are available.
- How should the start year (reference year) and end year for the start-end contrast be chosen?

WGSAEM felt that the start-end contrast test is useful when the start-year corresponds to a fixed intervention point or a reference year. Then the start-end contrast provides a useful tool for assessing a reduction or increase in level. However, if the starting point is arbitrary and, e.g., the trend pattern is fluctuating, the estimate of the trend will also fluctuate. This may happen even for short time series.

There was some discussion on the proposed methods for assessing whether and how annual data should be adjusted. It was proposed to complement the methods by suitable scatter and residual plots.

6.3.3 IMPACT: extracting anthropogenic signals from environmental data influenced by meteorological covariates

The main objective of the EU-funded project IMPACT was to develop new analytical tools that can facilitate the extraction of anthropogenic signals from environmental data influenced by meteorological conditions and other sources of natural variation.

Three aspects were considered:

1) Incorporation of covariates (indicators of natural variation) into Mann-Kendall tests for monotone trends;
2) Flow-adjustment of riverine loads of nutrients;
3) Incorporation of deterministic models into statistical procedures.

Incorporation of covariates into Mann-Kendall tests

It is widely recognized that meteorologically driven variation or other types of natural variation in environmental quality data can long conceal or distort important anthropogenic trends. In principle, there are two different ways to take this into account in trend assessments:

1) meteorological adjustment of collected data followed by trend analysis of adjusted data;
2) simultaneous adjustment and trend analysis.

Multivariate regression models may represent the second approach. However, non-parametric procedures can also be used. A so-called Partial Mann-Kendall (PMK) test for trends in the presence of a covariate can be based on the test statistic

$$Z = \frac{T_1 - \hat{\rho}\dfrac{\sigma_2}{\sigma_1}T_2}{\sqrt{(1-(\hat{\rho})^2)}\sigma_1}$$

where T_1 and T_2 denote the test statistics for the response variable and the covariate, respectively, σ_1 and σ_2 are the standard deviations for each of the test statistics, and ρ is the correlation between T_1 and T_2 (see Libiseller and Grimvall (2002) for precise details). The PMK test is primarily intended for situations in which a good adjustment method is difficult to construct or the adjustment causes an undesirable smoothing of the true random variation in the response variable. A variety of different PMK tests for data collected over several seasons and possibly also over several sites is presented in Libiseller and Grimvall (2002). Software for the trend assessment can be downloaded from the IMPACT website (www.mai.liu.se/impact.html).

Flow-adjustment of riverine loads of nutrients

The relationship between riverine loads and water discharge often has a pronounced seasonal variation. If there is a significant long-term trend in the data, such relationships may also change gradually from year to year. This calls for flow-adjustment models in which the parameters vary with both season and year. A semi-parametric procedure, originally developed by Stålnacke and Grimvall (2001), was extended to accommodate several covariates and tested on data from the lower reaches of the Elbe River. The basic structure of the model is a linear relationship between load and flow, where the intercept and slope parameters are time-dependent. The intercept is treated in a non-parametric fashion, and the slope parameters are season-specific but identical for all years during the study period. The study of Elbe data showed that, after adjustment, the annual loads of nitrogen and phosphorus formed a smooth curve that greatly facilitated the assessment of progress towards environmental targets, such as the 50% reduction target. The theoretical formulation of the model and details of the Elbe study can be found in a report by Hussian *et al.* (2003). Software for the normalization can be downloaded from the IMPACT website (www.mai.liu.se/impact.html).

Incorporation of deterministic models into statistical normalization procedures

Trend assessments of environmental quality data can often be improved by incorporating elements of process-based deterministic models into statistical normalization (adjustment) techniques. A case study of the deposition of wet nitrate at a sampling site in Sweden was presented. Analysis of model inputs and outputs helped to reveal the meteorological variables that had the strongest impact on the wet nitrate deposition. In addition, it was found that most of the variation in monthly wet deposition data could be explained by a relatively simple linear regression model involving the amount of precipitation and the number of precipitation days by sector values derived from back-trajectories. This reduced model was then used to meteorologically normalize measured wet deposition data.

It can be noted that process-based models have two different roles in the procedure just described:

- They helped to identify important explanatory variables.
- Model-computed sector values for back-trajectories of the sampled precipitation were used as explanatory variables.

The case study of wet nitrate deposition can be found in Hallberg and Grimvall (2002). Case studies of tropospheric ozone are reported in Libiseller *et al.* (2003) and Libiseller and Grimvall (2003).

References

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Hallberg, C., and Grimvall, A. 2002. Meteorological normalisation of wet nitrate deposition. (e-mail: angri@mai.liu.se).

Libiseller, C., and Grimvall, A. 2002. Performance of partial Mann-Kendall tests for trend detection in the presence of covariates. Environmetrics, 13: 71–84.

Libiseller, C., and Grimvall, A. 2003. Model selection for local and regional approaches to meteorological normalisation of background concentrations of tropospheric ozone. Atmospheric Environment, 37: 3923–3931.

Libiseller, C., Grimvall, A., Waldén, J., and Paatero, J. 2003. Meteorological normalisation of tropospheric ozone using back-trajectories. Submitted to Atmospheric Environment. (e-mail: cllib@mai.liu.se).

Stålnacke, P., and Grimvall, A. 2001. Semiparametric approaches to flow-normalisation and source apportionment of substance transport in rivers. Environmetrics, 12: 233–250.

6.4 Statistical Aspects in Relation to the Development of Environmental Indicators

Request

This is part of continuing ICES work in relation to statistical aspects relevant to the development of environmental indicators.

Source of the information presented

The 2003 report of the Working Group on Statistical Aspects of Environmental Monitoring (WGSAEM) and ACME deliberations.

Summary

The work done for the European Environment Agency (EEA) on compiling data and assessing time trends and recent status of hazardous substances in the marine environment through ETC/WTR (European Topic Centre for Water) was discussed by WGSAEM. The work has been carried out by the Norwegian Institute for Water Research (NIVA) as one of the marine partners within ETC/WTR. During 2002, fact sheets were produced based on available data collected from ICES and the Conventions (HELCOM, OSPAR) and from individual countries according to preliminary guidelines. The fact sheets form the basis for the EEA environmental indicator reports.

The work revealed a number of problems with data reporting, with implications for how environmental assessment can be carried out. These problems have been pointed out in the fact sheets, and can be summarized as follows.

Many of the individually reported data sets had problems that made it inadvisable to use them. Common problems were:

- Notation that data are below the detection limit, but the detection limit was not given;
- Notation that data are below or above a management level, but the management level was not given; e.g., data were reported as "above acceptable level" or "below acceptable level", with no definition of the acceptable level;
- Data were revealed by frequency patterns to be observations below the detection limit, but were reported as equal to the detection limit without any less-than flag;
- Concentration basis (e.g., wet weight or dry weight) was not given;
- Basis ratios were not given (e.g., % wet weight), prohibiting conversion to another basis;
- Submission was made of aggregated values without indication as to how these were derived nor what they represent in terms of spatial and temporal variability;
- Poor sample identification;

Table 6.4.1. Number of stations included in the indicator calculations of concentrations of cadmium over selected species, sea regions, and countries by year.

	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999
MUSSELS															
NE Atlantic															
Belgium		1		1	1	1	1	1	1	1	1	1		1	1
France		8	13	13	8	16	7	7	8	11	11	11	6	6	
Germany		5	4	1	5	8	2	9	7	9	3	3			
Iceland						3	9	9				9	11	9	7
Ireland								6	11	10	9	9			
The Netherlands		3	2	2	2	3	2	2	2	2	2	1	2	2	2
Norway		8	13	13	13	19	22	24	26	27	30	26	20	18	18
Spain						8	8	7	7	7	10	7	7	7	7
Sweden		1	2	2	2	2	2	2	1	2	2	1			
United Kingdom		1	1												
Mediterranean															
France	15	17	16	16	16	18	18	18	18	18	18	18	18	16	16
Greece									4	3	5	11	15	5	
COD															
Belgium		1	1	1	1		1	1	1	1	1	1		1	
The Netherlands		1		1											
Norway		2	4	4	4	7	7	7	6	9	11	8	8	8	8
Poland			3	3	1	2	2	1	1						
Sweden		2	2	2	2	2	2	2	2	2	2	1			

- Lack of information about monitoring purpose or type of monitoring station or indicator species;
- Time series in many cases were identifiable only by location coordinates, and not by proper station identification, resulting in uncertain assignment of data to time series in cases of minor coordinate changes between observation times.

Recommendations and advice

ICES recommends that Member Countries and the EEA allocate sufficient resources for the data collection, reporting, and collation processes, both in terms of calendar time and work effort, to provide a sound basis for statistical analysis and assessments in relation to the development and presentation of environmental indicators. In this work, the issues listed above should be taken into account.

Scientific background

Due to the various problems mentioned in the above summary, and also due to very variable amounts of data from different countries, the spatial and temporal coverage of acceptable data was very unbalanced and incomplete, as illustrated in Table 6.4.1. These problems make inferences from the data more uncertain, both in terms of their precision and in their representativeness of trends and spatial differences.

To improve the data collection process, a first version of formal guidelines for EEA reporting on marine waters has now been established. This is being applied in a new round of data collection, resulting in the production of revised fact sheets in the first part of 2003.

6.5 Methodology for Joint Assessments of Input Data and Data on Contaminants in Biota and Sediments

Request

Item 2.1 of the 2002 Work Programme from the OSPAR Commission: to participate in the joint assessment of concentration and input data to apply the trend assessment procedure.

Source of the information presented

The 2003 report of the Working Group on Statistical Aspects of Environmental Monitoring (WGSAEM) and ACME deliberations.

Summary

WGSAEM discussed different methods for joint assessments of trends and interrelationships between contaminant inputs and concentrations observed in monitored organisms, continuing on from the consideration of this topic in 2002 (ICES, 2003). Previous attempts by the OSPAR Working Group on Monitoring (MON) to identify a linear relationship between riverine and atmospheric inputs and concentrations in biota had been successful with some data sets, but had generally been unsuccessful.

One approach, presented at the OSPAR MON meeting in 2002 (OSPAR, 2002), considered the relationship between discharges of ^{99}Tc from British Nuclear Fuels, Sellafield, and the concentrations observed in the seaweed *Fucus*. It was demonstrated that this model was more successful to explain the relation between discharges and the concentrations measured in the *Fucus* compared to a simple correlation between the two variables. WGSAEM suggested that an additional parameter could be included in the model to estimate the accumulated discharge available to the *Fucus* when monitoring began. They also noted that the model could be expected to work less well with multiple or atmospheric sources, or with (less-well) estimated discharges, e.g., riverine inputs. Further, the model might be a special case of an ARIMA (Auto-Regressive Integrated Moving Average) model with an exogenous variable and encouraged further development of complex models.

Scientific background

Suppose $R_e(i - j - \delta)$ is the radioactivity discharged to the environment $(j - \delta)$ time-units prior to time-unit i. Then $R_m(i)$, the observed activity in a monitored organism in time-unit i, subject to sampling error, will be proportional to the sum of all previous discharges adjusted for radioactive decay and environmental loss (e.g., dispersion). Assuming that these losses cause historic radioactivity to follow a simple exponential decline, we can write

$$R_m(i) = a_{season} \sum_{j=i}^{\infty} R_e(i - j - \delta)k^j + \varepsilon_i$$

where

$$k = e^{-(\lambda_d + \lambda_e)}$$

and λ_d and λ_e are the radioactive and environmental decay rates respectively, α_{season} is a season-specific constant incorporating a concentration factor and scaling effects, δ is a lag effect reflecting distance from point of discharge, and ε_i is a random component representing sampling and measurement variability in the ith observation.

The data are shown in Figure 6.5.1. There is a linear correlation coefficient of 0.53 (28% of the variation explained) between the concentrations in *Fucus* from a site near to the point of discharge and the corresponding monthly discharges.

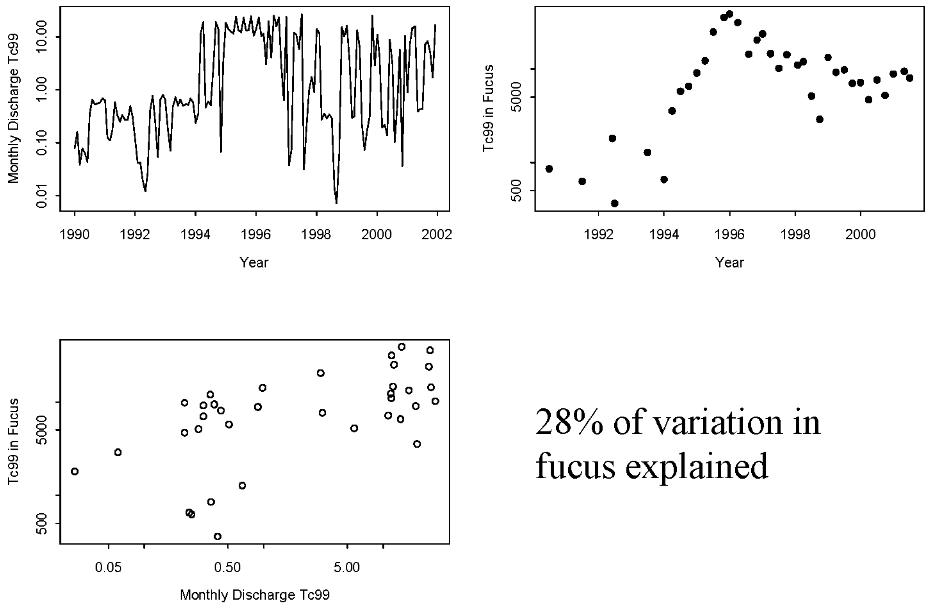
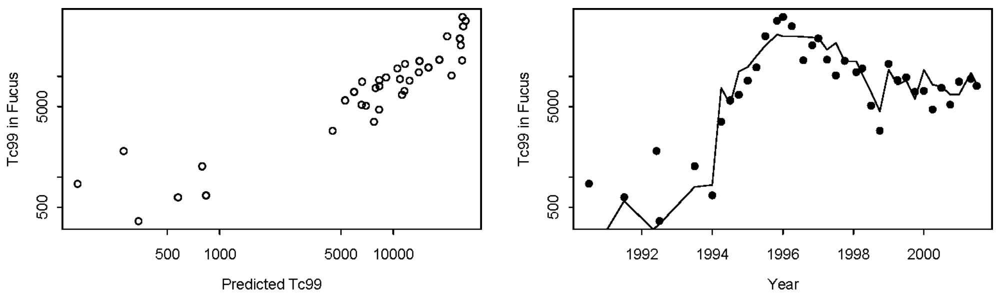


Figure 6.5.1. The monthly discharges of ^{99}Tc (top left), the irregular observations of concentrations in *Fucus* from a site near to the point of discharge (top right), and the concentrations in *Fucus* plotted against the corresponding monthly discharges (bottom left).



77% of variation in
fucus explained

Figure 6.5.2. Observed concentrations of ^{99}Tc in *Fucus* plotted against the concentrations predicted by the model with $\delta = 0$ and $k = 0.9$ (left) and the time series of observed concentrations together with the time series of predicted values (right).

The results of fitting the model with $\delta = 0$ and $k = 0.9$ are shown in Figure 6.5.2. The linear correlation coefficient between the observed concentrations in *Fucus* and those predicted from the model is now 0.88 (77% of the variation explained). The fit of the model to the data is reasonable, although somewhat poorer in the early years of the time series. The data and analysis are described in more detail in Namakowski *et al.* (in prep.).

References

ICES. 2003. Report of the ICES Advisory Committee on the Marine Environment, 2002. ICES Cooperative Research Report, 256: 32–34.

Namakowski, C., Nicholson, M.D., and Kershaw, P. (in prep.). Uptake by *Fucus* of technetium-99 discharged into the Irish Sea.

OSPAR. 2002. Joint assessments of contaminants in the marine environment. Paper presented at the 2002 meeting of the Working Group on Monitoring. MON 02/5/1-E. (available at http://www.ospar.org).

6.6 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 (http://www.ices.dk/status) as material becomes available. A summary of the 2002/2003 Environmental Status Report is provided here.

6.6.1 Oceanographic conditions

Source of the information presented

The 2003 report of the Working Group on Oceanic Hydrography (WGOH), the ICES Ocean Climate Status Summary 2002/2003, and ACME deliberations

Summary

In most areas of the North Atlantic, during 2002 temperature and salinity were higher than the long-term average. The North Atlantic Oscillation (NAO) index switched back to negative conditions during the winter preceding 2001, having recovered in the previous four years from the extreme negative value of 1996, which had brought to an end a period of extreme and persistent positive NAO index in the late 1980s/early 1990s. The 2002 NAO index showed a return to positive values which for the winter as a whole was not extreme, but individual months exhibited extreme and opposing sea-level pressure anomaly patterns.

The oceanographic conditions in the various ICES areas may be summarized as follows:

- Summer temperatures off West Greenland were slightly above average, but autumn temperatures were slightly below average. Unusually low salinities were observed in the off-slope surface waters during autumn.
- Annual mean air temperatures over all areas of the Northwest Atlantic remained slightly above normal, but have decreased from 2001 levels. The amount of sea ice on the eastern Canadian Continental Shelf was below normal for the seventh consecutive year. Ocean temperatures remained above normal, thus continuing the warm trend experienced throughout much of the Northwest Atlantic during the past several years. Ocean salinities increased to the highest levels observed in over a decade.
- The 2001/2002 Labrador Sea winter was more severe than the previous winter but still milder than normal. Observations in early summer 2002 showed remnants of convective overturning to maximum depths of 1200–1400 m, about 400 m deeper than seen in the preceding two years. Apart from the slight increase in winter convection, the general trend was to warmer and more saline conditions. This was true both in waters shallower than the maximum depth of convection and in the intermediate depths below 1400 m. The net result is that the mean 0–2000 m salinity was the highest in the past thirteen years of regular spring-summer observations. The corresponding mean temperature was the second highest observed during this period.
- Temperature and salinity during winter and spring were below the long-term mean on the shelf north, northeast, and east of Iceland. Summer and autumn values in this area were about average. The salinity and temperature of the Atlantic water originating from south of Iceland remained at high levels, similar to previous years though slightly lower than the peak values of 1998.
- Very cold weather in the spring-summer period in the southern Bay of Biscay area made 2002 the coldest sea surface temperature year since 1992. Upper water (0–300 m) mean temperatures were close to the long-term average. Salinity levels are now recovering following the minimum in 2001 and values at shallower depths are now average.
- On the WOCE/CLIVAR Section A1E, temperature and salinity were relatively high in the upper layer. This suggests that a new positive salinity anomaly is in progress. In the upper 1200 m of the water column, the tendency is towards warmer and more saline conditions. This is due to the deepening and decay of the Labrador Sea water mass produced in the 1990s.
- With respect to the last four decades, Atlantic waters in the Faroe-Shetland Channel are generally warming and becoming more saline. This trend is continuing.
- Surface water temperatures were higher than average in most areas for the whole of the year. Salinities in

the North Sea returned to normal following the extreme low values observed in 2001.

- Late summer was unusually warm, which resulted in higher than normal sea surface temperatures at that time. Low surface salinities were found in the Kattegat and Skagerrak in April–June due to large outflows from the Baltic Sea.
- The long-term warming trend continued, and the temperature of Atlantic water was the highest since the time series started in 1978.
- Temperatures were warmer than average. Positive temperature anomalies increased from average levels in January to a maximum in June, which was the highest of the last thirty years. Temperature anomalies then decreased to the long-term mean by the end of the year.
- The Atlantic waters of the West Spitsbergen Current were characterized by high temperature and salinity in summer, similar to those observed during the last three years. Polar waters in the east Greenland Current were significantly colder and less saline than in summer 2001.

Scientific background

Sea-level atmospheric pressures typically show a sub-polar low centered near Iceland and a sub-tropical high over the Azores. When the Icelandic Low deepens, the Azores High tends to strengthen and when the Low weakens so too does the High. This atmospheric variability is known as the North Atlantic Oscillation (NAO). A NAO index was defined based upon the wintertime pressure difference between Iceland and either the Azores or Lisbon, Portugal. Under a positive NAO phase (intensification of the pressure systems), there tends to be increased westerly winds across the Atlantic, northwesterly winds over the Labrador Sea, and southeasterly winds over northern Europe; an increase in significant wave heights across the Atlantic; anomalous cooling in the Labrador Sea and off North Africa, with warming over the southeastern United States seaboard and northern Europe and the Barents Sea; increased ice over northern areas where cooling occurs and reduced ice with warming; increased number of icebergs drifting onto the Grand Banks; an increase in the transport of the Gulf Stream with a corresponding northward shift of the Stream; decreased transport in the Labrador Current; increased convection in the Labrador Sea; and greater dust transport from North Africa across the sub-tropical Atlantic (Figure 6.6.1.1). A negative phase (weakening of the pressure systems) produces the opposite response. The NAO index varies greatly from year to year but with discernible longer-period trends. This includes significant intensification from a minimum in the 1960s to high values in the 1990s superimposed upon strong decadal fluctuations. However, the close relation that is found between NAO and certain physical and biological parameters does not seem to be universal. In certain periods the relation breaks down, and when the NAO in 1996 suddenly declined to low values after many years above normal values, the recovery of the previous circulation pattern varied across the Atlantic (ICES Climate Status Report 1999/2000; http://www.ices.dk/status/clim9900).

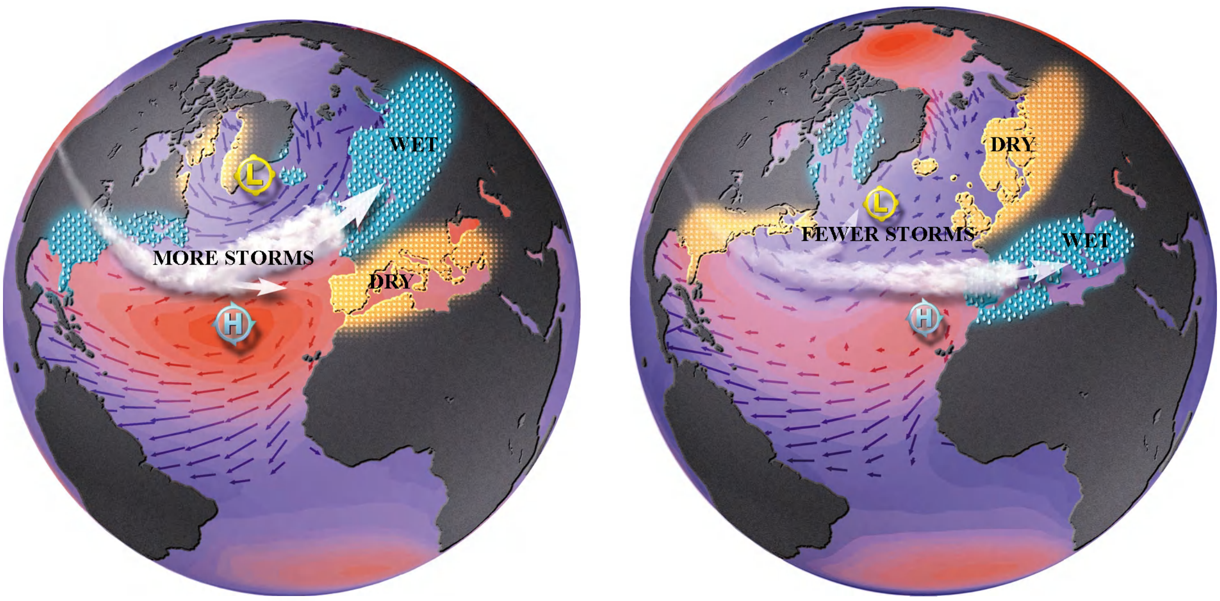


Figure 6.6.1.1. Positive NAO index (left) and negative NAO index (right). Source for figure: Martin Visbeck, Columbia University.

6.6.2 Plankton monitoring results

Source of the information presented

The 2003 reports of the Working Group on Phytoplankton Ecology (WGPE) and the Working Group on Zooplankton Ecology (WGZE), and ACME deliberations.

Summary

In its report, WGPE produced a phytoplankton status report which provides detailed indications of the composition of phytoplankton for the year 2001, month by month. This information was prepared by Germany (North Sea and Baltic areas), the Netherlands, and Sweden (Skagerrak, Kattegat, southern and central Baltic Sea). These numerous taxonomic indications are difficult to synthesize, and the trends in phytoplankton for the year are not easily revealed. However, an example is given of the phytoplankton trends for the Helgoland-Reede station. The phytoplankton status report will be available on the ICES Environmental Status web page: http://www.ices.dk/status/.

The zooplankton status report will also be available on the ICES Environmental Status web page: http://www.ices.dk/status/. Collections were updated to December 2002, and the metadata tables were improved. In addition, tables on the annual variability of the top ten zooplankton species were added for Plymouth and Helgoland. Also, seasonal and year-to-year variability was given for two target species: *Acartia clausi* and *Calanus helgolandicus*. The regional coverage of the report includes five fixed stations and 27 standard sections, in both North America and Europe, from the south of Portugal to northern Norway.

Scientific background

Important information is given in the WGZE report on the sampling devices, the periodicity of sampling, and the different data collected, which will be helpful in interpreting the results of these monitoring programmes.

Figure 6.6.2.1 shows the zooplankton sampling network in the ICES area, covering those sampling programmes reported to the WGZE.

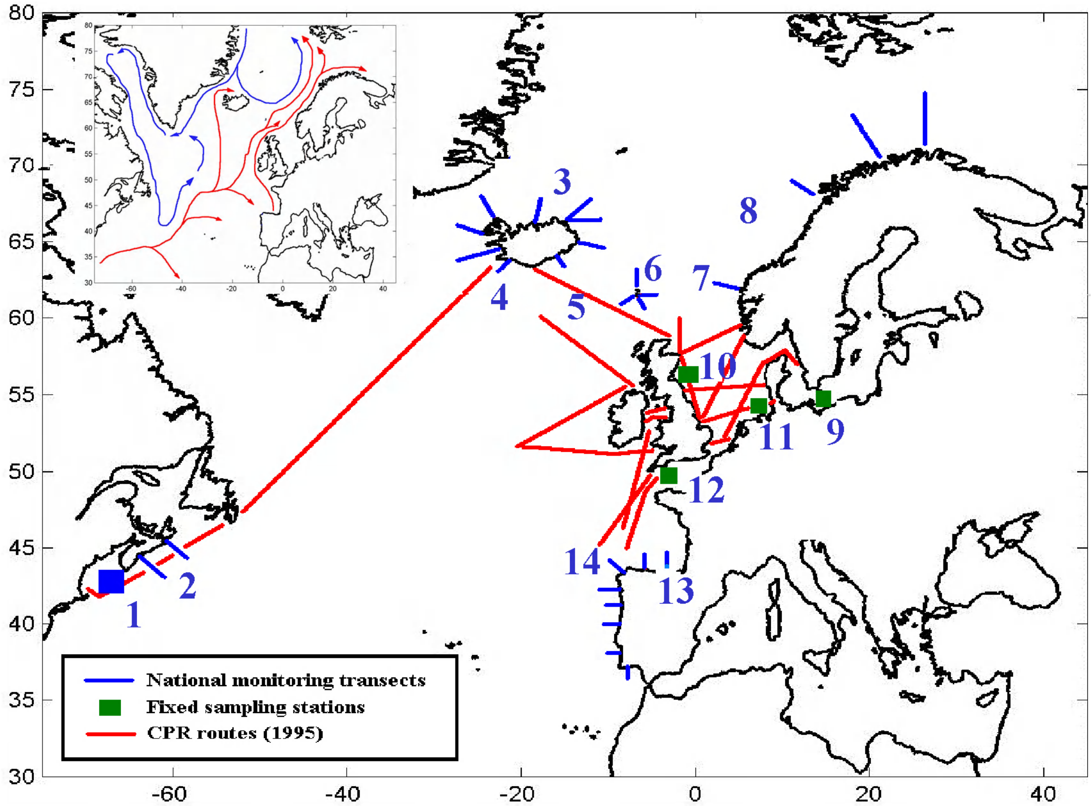


Figure 6.6.2.1. Zooplankton sampling network in the ICES area (only covering sampling programmes reported to the WGZE); numbers refer to the area designations used in this report. The map in the upper-left corner represents the schematic general circulation in the North Atlantic.

Area 1: Georges Bank

Two examples of interannual variations are presented here, showing the plankton displacement volume on Georges Bank in the early spring and early autumn (Figure 6.6.2.2). The trend observed reveals a slow decline in the plankton volume over a 28-year period.

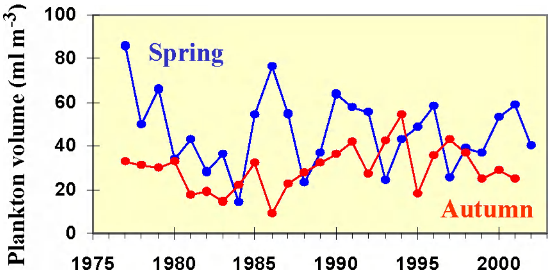


Figure 6.6.2.2. Plankton displacement volume on Georges Bank in the early spring and early autumn.

Area 2: Emerald Basin (West Atlantic, Scotian Shelf)

A stock status report on the state of the phytoplankton and zooplankton in Canadian Atlantic waters is prepared every year. This report is also published on the web at:

http://www.dfo-mpo.gc.ca/csas/Csas/English/Status/general.htm.

The *C. finmarchicus* population declined between 1995 and 1997 to reach the historical low levels of 1984. During 1998 and 1999, the population had recovered reaching maximum levels in the autumn of 1999, but again populations declined to a low in 2002. *C. finmarchicus* accounts for a significant portion of total zooplankton, which shows the same general pattern in abundance.

Areas 3 and 4: Icelandic waters

The zooplankton biomass north of Iceland is influenced by the inflow of warm Atlantic Water to the area. Thus, in warm years, when the flow of Atlantic Water onto the northern shelf is high, the zooplankton biomass is almost two times higher than in cold years, when this inflow is not as evident.

Comparison with other data from the northern North Atlantic shows that the zooplankton biomass observed in spring is descriptive of the mean copepod biomass in that year. Recent research also shows that the variation of zooplankton biomass in the Icelandic area is in correspondence with the long-term variability of zooplankton abundance over a much larger area, i.e., in the northern North Atlantic in general.

Area 5: Iceland-Scotland CPR line

The series is for total copepods along a Continuous Plankton Recorder (CPR) route between the north of Scotland and Iceland during the period 1958–2001. In the Scotland-Iceland area, the mean total copepod abundance in 2001 was 197.8 individuals per sample (~3 m^3), almost half below the overall mean for the series of 344.7 individuals per sample (http://www.sahfos.org).

Acartia spp. is the most important species in terms of abundance, accounting for 41% of the total abundance of copepods. Year-to-year variability of this species is shown. *Acartia* spp. abundance in 2001 was 91.6 ind per 3 m^3, which is one third below the annual average (142.8 ind per 3 m^3). The decreasing trend evidenced in the time series mirrors the trend obtained for the total copepods.

Area 6: Faroe Islands

Data on the average zooplankton biomass in the upper 50 m in the two water masses in the oceanic part of the section are available in May for the period 1990–2002 with few gaps. Values of abundance in 2002 were in the average of the long-term mean. *C. finmarchicus* is the dominant species in both water masses. With the exception of 1993, the biomass was clearly higher in the cold-water mass in the northern part of the section than in the warmer southern part. The reason is a higher abundance of overwintered *C. finmarchicus* (CV and adults), together with some *Calanus hyperboreus* in the northern part. In the Atlantic water, many fewer large individuals are present, but there is a larger number of small (recruit) stages in May. Since reproduction starts earlier in the southern part of the section, the total numbers of *C. finmarchicus* are higher on average in the Atlantic water than in the East Icelandic Current Water, despite the lower biomass.

Areas 7 and 8: Svinøy and the Norwegian Sea

The development of zooplankton biomass in spring at the Svinøy transect showed very small variations among years in the period 1997–2001 and the maximum biomass in early summer varied from 8 to 9.3 g DW m^{-2}. In 2002, the maximum biomass as an average for all stations was 11.32 g DW m^{-2} (28–30 April), higher than in previous years. In April, the highest biomasses were observed at the three innermost shelf stations (11.7–18.8 g DW m^{-2}), and at the five westernmost deep-ocean stations (9.95–30.22 g DW m^{-2}). The remaining stations, both at the shelf and the deep ocean, showed rather low biomasses, varying between 1.94 and 8.3 g DW m^{-2}. In July 2001 there was a clear trend towards higher biomasses at the ten easternmost stations.

Due to the still reduced coverage of the Svinøy transect (four and five times in 2001 and 2002, respectively), no firm conclusions can be made in relation to previous years with regard to the zooplankton development.

Area 9: Arkona Basin (Germany)

The Baltic Monitoring Programme (BMP) consists of 24 international stations. One station (54°55'N, 13°30'E) has been chosen from the database. In some years, the sampling coverage is quite poor (e.g., 1995 and 1996). Variations in the range 10,000–50,000 ind m^{-3} are typically observed during the seasonal cycle in the southwestern Baltic Sea. Peaks of plankton observed in spring 1983, 1988, 1995, 1998, 2000, and 2002 occurred because of mass developments of rotifers, which often happens after mild winters. In spite of these peaks, the cladoceran *Bosmina coregonii* is the dominant species during summer when the water temperature reaches 16 °C. Although no statistical trend is observed, four of the six spring peaks mentioned above occurred during the last eight years.

Area 10: Stonehaven (Scotland, northwestern North Sea)

In the late summer and through the autumn of most years, water with a high Atlantic Ocean content passes down the Scottish east coast. These events are particularly observable in the salinity signal. For example, 1997 showed a strong salinity increase in the late summer, whereas 1998 showed very little. These influxes often bring oceanic species; for example, the chaetognath *Sagitta serratodentata* and the siphonophore *Muggiea atlantica* are indicators of this oceanic influence.

First and most abundant in the spring and summer is *C. finmarchicus*, an important species in that the large spring influx and production provide food for fish larvae in spring. However, its congener *C. helgolandicus*, a more southern species and generally most abundant in summer and autumn, has increasingly shown evidence of increased productivity and extended survival through the winter months. This is most likely a reflection of changes in the physical environment through the last few months of the year, with faster or slower cooling of the sea affecting the strongly temperature-dependent physiology of these small plankton.

Area 11: Helgoland (southeastern North Sea)

Acartia clausi represents a significant fraction of the total calanoid copepods. Their abundances were compared against the mean weekly abundance in the twenty-year time series 1975–1994. 2002 showed an earlier start in the development of small calanoid copepod populations (e.g., *A. clausi*) and a lower abundance than the average of the time series. The population of the warm-water cladoceran *Penilia avirostris* was abundant again for the fourth consecutive year and must now be regarded as a neozoan member of the community of the German Bight. In the meroplankton, echinoderm larvae (Asteroidea, Ophiuroidea, and Spatangoidea) had an earlier start of season and an increased autumnal abundance.

Area 12: Plymouth (Channel)

The ten dominant taxa at the station off Plymouth have been ranked according to their annual mean proportion of the total zooplankton. Over the time series, *Pseudocalanus* has been the most abundant, making up 12% of the total population. In 2002 its population increased by 40% compared to the yearly average. The *Temora* population for this year doubled over that of its yearly average levels. The *Oncaea* population had the highest decrease in 2002, of 64%; this was closely followed by *Evadne,* with a 63% decrease. The only other species in the top ten with a decreased population is that of *Appendicularia,* with a decrease of 34%.

Zooplankton abundance at the station off Plymouth showed a decreasing trend from 1988 to 1995, but increased somewhat from then until 1999. High variability can be seen in some years; for example, in 1990 for *Acartia clausi* and in 2002 for *Calanus helgolandicus*, the abundance was more than double that of the 90th percentile. A period of low abundance can be observed for both species during the years 1994, 1995, and 1996; this mirrors the low abundance of the total zooplankton at this time.

Area 13: Santander (southern Bay of Biscay)

Long-term changes of zooplankton abundance at Santander show a slight decreasing trend. This result is in opposition to the upward trend shown by the water column stratification index. This relationship between zooplankton and environmental conditions highlights the importance that the longer duration of the water column remaining stratified could have in limiting the interchange of nutrients from deeper to surface waters and, consequently, limiting the growth of phytoplankton and zooplankton.

Area 14: La Coruña (northwest Iberian Peninsula)

Zooplankton values in La Coruña differ from those in Santander. The zooplankton abundance is higher in La Coruña and the time series does not show any trend. Both characteristics are partly due to the influence of the seasonal upwelling, which prevents the water column from properly stratifying, reinforces the input of nutrients to the photic layer, enhances the growth of phytoplankton populations and, therefore, enhances the growth of zooplankton populations.

6.6.3 Harmful algal blooms

Source of the information presented

The 2003 report of the ICES-IOC Working Group on Harmful Algal Bloom Dynamics (WGHABD) and ACME deliberations.

Summary

WGHABD maintains an overview of the occurrence of harmful algal bloom events, and has developed decadal maps of harmful bloom events, which are included in the ICES Environmental Status Report at http://www.ices.dk/status/decadal/. These maps cover the following types of events: amnesic shellfish poisoning (ASP), ciguatera fish poisoning (CFP), diarrhetic shellfish poisoning (DSP), neurotoxic shellfish poisoning (NSP), paralytic shellfish poisoning (PSP), and cyanobacterial toxin poisoning.

Studies carried out in the Baltic Sea have pointed out the deleterious effects of cyanobacterial blooms on the survival and fecundity of wild fish. This finding is significant, since it may indicate that the most important effect of these large blooms may occur at the ecosystem level.

WGHABD has planned intersessional work to define regions for the Harmful Algal Event Database, HAEDAT, in order to automatically update the HAEDAT maps.

6.6.4 Fish disease prevalence

Source of the information presented

The 2003 report of the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) and ACME deliberations.

Summary

Only limited progress was achieved in the past year regarding the updating of the web-based report on diseases of wild and farmed marine fish and shellfish as part of the ICES Environmental Status Report. This was largely caused by an inadequate amount of current disease data being submitted to the ICES Marine Data Centre by Member Countries.

Recommendations and advice

ICES recommends that Member Countries gathering fish disease data submit these data to the ICES Marine Data Centre in order to update the web-based report on diseases of wild and farmed marine fish and shellfish as part of the ICES Environmental Status Report.

Scientific background

At its 2003 meeting, WGPDMO reviewed progress regarding the web-based report on diseases of wild and farmed marine fish and shellfish as part of the ICES Environmental Status Report. The most recent report is available at: http://www.ices.dk/status/fish_and_shellfish_diseases/index.htm.

Note was taken of the limited progress achieved in the update of the information and the development of current trend figures, which was largely caused by an inadequate amount of current data being submitted to the fish disease database of the ICES Marine Data Centre by Member Countries.

The ACME endorsed the recommendation made by WGPDMO that work on this project be continued. Issues to be addressed until the 2004 WGPDMO meeting concern the following:

- Update of the trend estimations using current disease data;
- Establishment of criteria for the inclusion of data in the trend maps;
- Development of a colour coding system for disease prevalence data;
- Development of maps providing relative prevalence levels;
- Separation of winter and spring data by adjustments made to the trend estimation procedures.

A progress report will be presented and reviewed at the 2004 WGPDMO meeting and will be forwarded to the ACME.

7 MARINE CONTAMINANTS: DISTRIBUTION, TRANSPORT, AND EFFECTS

7.1 Availability of Analytical Methods for Determination of Environmental Concentrations or Effects of substances on the OSPAR List of Chemicals for Priority Action

Request

Item 1 of the 2003 Work Programme from the OSPAR Commission: to provide advice on whether there are suitable analytical methods available to allow the measurement of environmental concentrations or effects of the substances listed on the OSPAR list of chemicals for priority action (as amended by OSPAR 2002), and whether any information exists on the presence of these chemicals in the marine environment.

Source of the information presented

The 2003 reports of the Marine Chemistry Working Group (MCWG) and the Working Group on Biological Effects of Contaminants (WGBEC), and ACME deliberations.

Summary

Updated information is provided on the status of analytical methods for determining environmental concentrations of substances on the OSPAR List of Chemicals for Priority Action, including whether or not there is knowledge of concentrations of the substances in the environment (Table 7.1.1). Similarly, the status of methods for determining the biological effects of these substances on samples collected in the field is also provided (Table 7.1.2).

With respect to the biological effects of these substances, some groups of substances have more than one biological effect and some groups of substances are heterogeneous. Few of the methods are specific to one contaminant, but will rather respond to a range of substances. Only methods that can be applied to field-collected organisms have been included. Additional methods can be added if bioassays are considered. It is also important to note that responses to all the contaminants listed depend on the dose. Data have only been included where an environmentally relevant dose may cause effects.

Of the metals listed, good methods exist for determining concentrations of cadmium, inorganic mercury, lead/organic lead compounds, and organotin compounds. There are no good methods for organic mercury. Alkylated phenols are oestrogenic and there are good methods to detect such effects in the marine environment. There are also well-established methods to detect effects of PAHs, PCBs, dioxins, and dibenzofurans in marine ecosystems. As indicated in Table 7.1.2, many of the substances are known to be endocrine disruptors, but there may not be a technique to monitor substance-specific effects in the marine environment. Impacts from some of these may be detected through the use of bioassays.

For the brominated diphenylethers, chloronaphthalenes, non-*ortho* chlorobiphenyls and mono-*ortho* chlorobiphenyls, dioxins and furans, and PCBs, a number of recent references have been provided for both the analytical methodology and environmental distribution of these compounds. A list of overviews and review notes prepared by ICES since 1990 has also been provided. More detailed information on specific substances can be provided by the preparation of additional review notes, if required. Tables 7.1.1 and 7.1.2 should be updated on a regular basis since new information is continuously becoming available on the determination of these substances in environmental samples and the measurement of biological effects.

Recommendations and advice

ICES recommends that OSPAR take into account the information presented here for the substances listed in the implementation of the revised Joint Assessment and Monitoring Programme (JAMP) and the development of guidelines.

Scientific background

There are two parts to this request; the first is for analytical methods for the determination of environmental concentrations of substances on the OSPAR List of Chemicals for Priority Action and the second is for information on the biological effects of these substances. This information is provided in Tables 7.1.1 and 7.1.2. It should be noted that new methods are being continuously developed and validated and, therefore, these tables will need to be updated annually by MCWG and WGBEC.

If OSPAR requires review notes to be prepared for specific compounds, these can be prepared on request, but a number of the well-established determinands have already been the subjects of overviews or review notes and these are listed below.

Tables 7.1.1 and 7.1.2 also indicate the status of the analytical or biological effects methods using the following notation:

- fully validated methods are available and widely used (3);
- methods have been published, but have not been used widely (2);

- there is evidence of effects in experimental studies; some expert laboratories can undertake these analyses (1); or
- there is no or very limited knowledge (0).

Table 7.1.1 also indicates whether or not information on the environmental occurrence and distribution of the substance is available, as noted by (+).

Table 7.1.1. Information on the availability of suitable analytical methods to allow the measurement of environmental concentrations of the substances on the OSPAR List of Chemicals for Priority Action, and whether information has been reported on concentrations of these substances in the marine environment.

Chemical (CAS number and/or name)	**Analysis**[a]	**Presence**[b]	**Comment**
85–22–3 Benzene, pentabromoethyl	1	−	
36065–30–2 Benzene, 1,3,5-tribromo-2-(2,3-dibromo-2-methylpropoxy)	1	−	
732–26–3 2,4,6-tri-*tert*-butylphenol (phenol, 2,4,6-*tris*(1,1-dimethylethyl))	1	−	Blank problems
98–51–1 4-*tert*-butyltoluene (benzene, 1-(1,1-dimethylethyl)-4-methyl-)	1	−	
Brominated flame retardants	3	+	For penta-mix BDE congeners
	1	+	For octa-mix BDE congeners
	1	+	For BDE209
	1	+	For HBCD and polybrominated biphenyls (PBBs)
77–47–4 1,3-cyclopentadiene, 1,2,3,4,5,5-hexachloro-	2	+	
Certain phthalates (dibutylphthalate and diethylhexylphthalate)	2	+	Blank problems
115–32–2 Dicofol (benzenemethanol, 4-chloro-α-(4-chlorophenyl)-α-(trichloromethyl)-)	2	+	
115–29–7 Endosulphan (6,9-methano-2,4,3-benzodioxathiepin, 6,7,8,9,10,10-hexachloro-1,5,5a,6,9,9a-hexahydro-,3-oxide)	2	+	
2104–64–5 EPN (phosphonothioic acid, phenyl-, O-ethyl O-(4-nitrophenyl) ester)	0	−	
70124–77–5 Flucythrinate (benzene acetic acid, 4-(difluoromethoxy)-α-(1-methylethyl)-, cyano(3-phenoxyphenyl)methyl ester)	0	−	
28680–45–7 and 2440-02-0 Heptachloronorbornene (bicyclo[2.2.1]hept-2-ene, heptachloro-)	3	+	For some congeners (P26, P50, P62)
	1	+	For other toxaphene congeners
Hexachlorocyclohexane (HCH) isomers	3	+	For α-, β- and γ-isomers
107–46–0 HMDS (disiloxane, hexamethyl-)	1	−	
465–73–6 Isodrin (1,4:5,8-dimethanonaphthalene, 1,2,3,4,10,10-hexachloro-1,4,4a,5,8,8a-hexahydro-, (1.α.,4.α.,4a. β.,5. β.,8. β.,8a. β.)-)	2	−	Environmental levels very low
Organic lead compounds	2	+	For tetra-alkyl lead

[a]MCWG believes that (3) there are good validated methods available; (2) there are methods described in the literature; (1) it is possible for some laboratories to analyse the compound; or (0) no methods are known to the group.

[b]MCWG knows that there are environmental levels reported (+), or MCWG members do not have knowledge of any environmental data (−).

Table 7.1.1. Continued.

Chemical (CAS number and/or name)	Analysis[a]	Presence[b]	Comment
72–43–5 Methoxychlor (benzene,1,1'-(2,2,2-trichloroethylidene)bis(4-methoxy)	3	+	
Musk xylene	2	+	
Chlorinated naphthalenes	2	+	Individual isomers should be determined
51000–52–3 Neodecanoic acid, ethenyl ester	0	−	
Nonylphenol/ethoxylates (NP/NPEs) and related substances	2	+	
140–66–9 Octylphenol (phenol, 4-(1,1,3,3,tetramethylbutyl)-)	2	+	
Organic tin compounds	3	+	For trialkyl- and triphenyltin
1825–21–4 Pentachloroanisole	2	+	
Pentachlorophenol (PCP)	3	+	
603–35–0 Phosphine, triphenyl-	0	−	
Polycyclic aromatic hydrocarbons (PAHs)	3	+	
Polychlorinated biphenyls (PCBs)	3	+	
Polychlorinated dibenzodioxins (PCDDs), polychlorinated dibenzofurans (PCDFs)	3	+	
Short-chained chlorinated paraffins (SCCP)	1	+	
79–94–7 TBBA (phenol, 4,4'-(1-methylethylidene)bis[2,6-dibromo]-)	1	+	
2227–13–6 Tetrasul (benzene, 1,2,4-trichloro-5-[(4-chlorophenyl)thio]-)	2	+	
87–61–6 Trichlorobenzene (benzene, 1,2,3-trichloro-)	2	+	
120–82–1 1,2,4-trichlorobenzene (benzene, 1,2,4-trichloro-)	2	+	
108–70–3 1,3,5-trichlorobenzene (benzene, 1,3,5-trichloro-)	2	+	
55525–54–7 Urea, N,N'-bis[(5-isocyanato-1,3,3-trimethylcyclohexyl)methyl]-	0	−	
Cadmium	3	+	
Lead	3	+	
Mercury	3	+	
Organic mercury compounds	2–3	+	
Organic tin compounds	2–3	+	

[a]MCWG believes that (3) there are good validated methods available; (2) there are methods described in the literature; (1) it is possible for some laboratories to analyse the compound; or (0) no methods are known to the group.

[b]MCWG knows that there are environmental levels reported (+), or MCWG members do not have knowledge of any environmental data (−).

References

Recent references relating to the analysis and environmental distribution of the brominated diphenylethers:

de Boer, J., Allchin, C., Law, R., Zegers, B., and Boon, J.P. 2001. Method for the analysis of polybrominated diphenylethers in sediments and biota. Trends in Analytical Chemistry, 20: 591–599.

Law, R.J., Allchin, C.R., Bennett, M.E., Morris, S., and Rogan, E. 2002. Polybrominated diphenyl ethers in two species of marine top predators from England and Wales. Chemosphere, 46: 673–681.

Boon, J.P., Lewis, W.E., Tjoen-A-Choy, M.R., Allchin, C.R., Law, R.J., de Boer, J., ten Hallers-Tjabbes, C.C., and Zegers, B.N. 2002. Levels of polybrominated diphenyl ether (PBDE) flame retardants in animals representing different trophic levels of the North Sea food web. Environmental Science and Technology, 36: 4025–4032.

A total of 110 papers from platform and poster presentations at Dioxin 2003 provided information on flame retardants. The entire Volume 61 of Organohalogen Compounds is relevant to OSPAR and HELCOM. A selection of these papers is listed below:

Allchin, C.R., and Morris, S. 2003. Hexabromocyclododecane (HBCD) diastereoisomers and brominated diphenyl ether congener (BDE) residues in edible fish from the Rivers Skerne and Tees, UK. Organohalogen Compounds, 61: 41–44.

Kierkegaard, A., and Bjorklund, J. 2003. The presence of a "new" flame retardant, decabromodiphenyl ethane, in environmental samples. Organohalogen Compounds, 61: 183–186.

Letcher, R. *et al.* 2003. Polybrominated diphenyl ethers and hydroxylated and methoxylated analogues in Detroit River fish. Organohalogen Compounds, 61: 29–32.

Morris, S. *et al.* 2003. A new LC-MS method for the detection and quantification of hexabromocyclododecane diastereoisomers and tetrabromobisphenol-A flame retardants in environmental samples. Organohalogen Compounds, 60: 436–439.

Muir, D. *et al.* 2003. Current deposition and historical profiles of decabromodiphenyl ether in sediment cores. Organohalogen Compounds, 61: 77–80

Tomy, G. *et al.* 2003. Congener-specific analysis of hexabromocyclododecane (HBCDD) by high performance liquid chromatography electrospray tandem mass spectrometry. Organohalogen Compounds, 60: 448–451.

She, J. *et al.* 2003. High PBDE levels in piscivorous seabird eggs from the San Francisco bay and Washington State. Organohalogen Compounds, 61: 33–36.

Stapleton, H.M. *et al.* 2003. Debromination of decabromodiphenyl ether by juvenile carp (Cyprinus carpio). Organohalogen Compounds, 61: 21–24.

Ueno, D. *et al.* 2003. Global pollution monitoring of polybrominated diphenyl ethers (PBDEs) using skipjack tuna as a bioindicator. Organohalogen Compounds, 61: 37–40.

In addition, there is a special issue of Environmental International on flame retardants, the contents of which are listed below:

Letcher, R.J., and Behnisch, P.A. 2003. The state-of-the-science and trends of brominated flame retardants in the environment: present knowledge and future directions. Environment International, 29(6): 663–664.

Watanabe, I., and Sakai, S.-i. 2003. Environment release and behavior of brominated flame retardants. Environment International, 29(6): 665–682.

Alaee, M., Arias, P., Sjodin, A., and Bergman, Å. 2003. An overview of commercially used brominated flame retardants, their applications, their use patterns in different countries/regions and possible modes of release. Environment International, 29(6): 683–689.

Alcock, R.E., Sweetman, A.J., Prevedouros, K., and Jones, K.C. 2003. Understanding levels and trends of BDE-47 in the UK and North America: an assessment of principal reservoirs and source inputs. Environment International, 29(6): 691–698.

Weber, R., and Kuch, B. 2003. Relevance of BFRs and thermal conditions on the formation pathways of brominated and brominated-chlorinated dibenzodioxins and dibenzofurans. Environment International, 29(6): 699–710.

Ebert, J., and Bahadir, M. 2003. Formation of PBDD/F from flame-retarded plastic materials under thermal stress. Environment International, 29(6): 711–716.

Gouin, T., and Harner, T. 2003. Modelling the environmental fate of the polybrominated diphenyl ethers. Environment International, 29(6): 717–724.

Santillo, D., and Johnston, P. 2003. Playing with fire: the global threat presented by brominated flame retardants justifies urgent substitution. Environment International, 29(6): 725–734

Covaci, A., Voorspoels, S., and de Boer, J. 2003. Determination of brominated flame retardants, with emphasis on polybrominated diphenyl ethers (PBDEs) in environmental and human samples—a review. Environment International, 29(6): 735–756.

Law, R.J., Alaee, M., Allchin, C.R., Boon, J.P., Lebeuf, M., Lepom, P., and Stern, G.A. 2003. Levels and trends of polybrominated diphenylethers and other brominated flame retardants in wildlife. Environment International, 29(6): 757–770.

Hale, R.C., Alaee, M., Manchester-Neesvig, J.B., Stapleton, H.M., and Ikonomou, M.G. 2003. Polybrominated diphenyl ether flame retardants in the North American environment. Environment International, 29(6): 771–779.

Kemmlein, S., Herzke, D., and Law, R.J. 2003. BFR-governmental testing programme. Environment International, 29(6): 781–792.

Hardy, M.L., Biesemeier, J., Manor, O., and Gentit, W. 2003. Industry-sponsored research on the potential health and environmental effects of selected brominated flame retardants. Environment International, 29(6): 793–799.
Hakk, H., and Letcher, R.J. 2003. Metabolism in the toxicokinetics and fate of brominated flame retardants—a review. Environment International, 29(6): 801–828.
Sjodin, A., Patterson, D.G., and Bergman, Å. 2003. A review on human exposure to brominated flame retardants—particularly polybrominated diphenyl ethers. Environment International, 29(6): 829–839.
Darnerud, P.O. 2003. Toxic effects of brominated flame retardants in man and in wildlife. Environment International, 29(6): 841–853.
Birnbaum, L.S., Staskal, D.F., and Diliberto, J.J. 2003. Health effects of polybrominated dibenzo-*p*-dioxins (PBDDs) and dibenzofurans (PBDFs). Environment International, 29(6): 855–860.
Behnisch, P.A., Hosoe, K., and Sakai, S.-i. 2003. Brominated dioxin-like compounds: *in vitro* assessment in comparison to classical dioxin-like compounds and other polyaromatic compounds. Environment International, 29(6): 861–877.
Legler, J., and Brouwer, A. 2003. Are brominated flame retardants endocrine disruptors? Environment International, 29(6): 879–885.

Some additional useful references on synthetic organic contaminants include:

de Boer, J., and Law, R.J. 2003. Developments in the use of chromatographic techniques in marine laboratories for the determination of halogenated contaminants and polycyclic aromatic hydrocarbons. Journal of Chromatography A, 1000(1/2): 223–251.
Clark, J.R., Goodman, L.R., Borthwick, P.W., Patrick J.M. jr., Cripe, G.M., Moody, P.M., Moore, J.C., and Lores, E.M. 1989. Toxicity of pyrethroids to marine invertebrates and fish: a literature review and test results with sediment-sorbed chemicals. Environmental Toxicology and Chemistry, 8: 393–401.
Muir, D.C.G., and Norstrom, R.J. 2000. Geographical differences and time trends of persistent organic pollutants in the Arctic. Toxicology Letters, 112/113: 93–101.
Rimkus, G.G. 1999. Polycyclic musk fragrances in the aquatic environment. Toxicology Letters, 111: 37–56.
Staples, C.A., Parkerton, T.F., and Peterson, D.R. 2000. A risk assessment of selected phthalate esters in North American and Western European surface waters. Chemosphere, 40: 885–891.
Sumpter, J.P. 2000. Endocrine disrupting chemicals in the aquatic environment. *In* Proceedings of the Sixth International Symposium on the Reproductive Physiology of Fish, Bergen, Norway, 1999, pp. 349–355. Ed. by B. Norberg, O.S. Kjesbu, G.L. Tarranged, E. Andersson, and S.O. Stefansson.
Voulvoulis, N., Scromshaw, M.D., and Lester, J.N. 2002. Comparative environmental assessment of biocides used in antifouling paints. Chemosphere, 47: 789–795.
Wade, T.L., Sericano, J.L., Gardinali, P.R., Wolff, G., and Chambers, L. 1999. NOAA's "Mussel Watch" Project: current use organic compounds in bivalves. Marine Pollution Bulletin, 37: 20–26.

ICES review notes and overviews are listed below, together with several other relevant references:

Hexachlorobenzene and lindane

ICES. 1990. Report of the ICES Advisory Committee on Marine Pollution, 1990. ICES Cooperative Research Report, 172: 82–85.

Toxaphene

ICES. 1994. Report of the ICES Advisory Committee on the Marine Environment, 1994. ICES Cooperative Research Report, 204: 58–59.
ICES. 1997. Report of the ICES Advisory Committee on the Marine Environment, 1997. ICES Cooperative Research Report, 222: 63–64, 141–149.
Vetter, W., and Oehme, M. 2000. Toxaphene. *In* The Handbook of Environmental Chemistry, Volume 3K, pp. 237–288. Ed. by J. Paasivirta. Springer-Verlag, Berlin. 2000. ISBN no. 3-540-65838-6.

Butyltin methodology

ICES. 1999. Report of the ICES Advisory Committee on the Marine Environment, 1998. ICES Cooperative Research Report, 233: 238–249.

Mercury

ICES. 1999. Report of the ICES Advisory Committee on the Marine Environment, 1998. ICES Cooperative Research Report, 233: 69, 263–274.

Synthetic musk compounds

ICES. 2000. Report of the ICES Advisory Committee on the Marine Environment, 1999. ICES Cooperative Research Report, 239: 88–89.

Brominated flame retardants (PBBs and PBDEs)

ICES. 2000. Report of the ICES Advisory Committee on the Marine Environment, 1999. ICES Cooperative Research Report, 239: 89, 169–192.

Tributyltin

ICES. 2000. Report of the ICES Advisory Committee on the Marine Environment, 1999. ICES Cooperative Research Report, 239: 88, 162–168.

Chloronaphthalenes

Jakobsson, E., and Asplund, L. 2000. Chloronaphthalenes. *In* The Handbook of Environmental Chemistry, Volume 3K, pp. 97–126. Ed. by J. Paasivirta. Springer-Verlag, Berlin. ISBN no. 3-540-65838-6.

Non-ortho and mono-ortho CBs

Kannan, N. 2000. Non-*ortho* and mono-*ortho* CBs. *In* The Handbook of Environmental Chemistry, Volume 3K, pp. 127–156. Ed. by J. Paasivirta. Springer Verlag, Berlin. ISBN no. 3-540-65838-6.

Dioxins and Furans

Liem, A.K.D., and Theelen, R.M.C. 1997. Dioxins: Exposure and Risk Assessment. Ph. D. thesis, University of Utrecht. ISBN 90-393-2012-8.

Oehme, M. 1998. Handbuch Dioxine. Spektrum Akademischer Verlag, Heidelberg. ISBN 3-8274-0126-7.

PCBs

de Boer, J. 2001. Environmental Analysis: Polychlorinated Biphenyls (chapter 7). *In* Handbook of Analytical Separations, Vol. 3, pp. 237–262. Ed. by W. Kleiböhmer. Elsevier Science B.V., Amsterdam.

Wells, D.E., and de Boer, J. 1999. Polychlorinated biphenyls, dioxins and other polyhalogenated hydrocarbons as environmental contaminants in food. *In* Environmental Contaminants in Food, pp. 305–363. Ed. by C.F. Moffat and K.J. Whittle. Sheffield Academic Press.

7.2 Information on Specific Contaminants

Request

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

Source of the information presented

The 2003 report of the Marine Chemistry Working Group (MCWG) and ACME deliberations.

Summary

The ACME took note of new information on several groups of marine contaminants, as prepared by the Marine Chemistry Working Group.

Table 7.1.2. Biological effects of substances on the OSPAR List of Chemicals for Priority Action. For each substance or group of substances, a number is given that indicates whether: there are available, widely used methods to monitor effects (3); methods have been published, but have not been widely used (2); there is evidence of effects in experimental studies (1); or there is no or very limited knowledge (0). The next column indicates whether there is knowledge that the component or metabolites of the component are endocrine disruptors (but does not indicate that there is a monitoring method). A question mark (?) next to a comment indicates that the effect is potential or uncertain.

Chemical (CAS number and/or name)	**Method**	**Endocrine disruptor**	**Comment**
85–22–3 Benzene, pentabromoethyl	0	not known	
36065–30–2 Benzene, 1,3,5-tribromo-2-(2,3-dibromo-2-methylpropoxy)	0	not known	
732–26–3 2,4,6-tri-*tert*-butylphenol (phenol, 2,4,6-*tris*(1,1-dimethylethyl))	3	yes	oestrogenic, vitellogenin induction
98–51–1 4-*tert*-butyltoluene (benzene, 1-(1,1-dimethylethyl)-4-methyl-)	2	yes	metabolite is oestrogenic
Brominated flame retardants	0–3	yes	some induce and some inhibit CYP1A; metabolites interact with thyroid metabolism
77–47–4 1,3-cyclopentadiene, 1,2,3,4,5,5-hexachloro-	0	not known	
Certain phthalates – dibutylphthalate	1	yes	weakly oestrogenic
Certain phthalates – diethylhexylphthalate	2	no	peroxisomal proliferator
115–32–2 Dicofol (benzenemethanol, 4-chloro-α-(4-chlorophenyl)-α-(trichloromethyl)-)	1	yes	similar mechanism as DDE; eggshell-thinning, affects reproduction of reptiles
115–29–7 Endosulphan (6,9-methano-2,4,3-benzodioxathiepin, 6,7,8,9,10,10-hexachloro-1,5,5a,6,9,9a-hexahydro-,3-oxide)	2?	yes	Endocrine-disrupting substance; high acute toxicity

Table 7.1.2. Continued.

Chemical (CAS number and/or name)	Method	Endocrine disruptor	Comment
2104–64–5 EPN (phosphonothioic acid, phenyl-, O-ethyl O-(4-nitrophenyl) ester)	0	not known	insecticide; not very persistent
70124–77–5 Flucythrinate (benzene acetic acid, 4-(difluoromethoxy)-α-(1-methylethyl)-, cyano(3-phenoxyphenyl)methyl ester)	0	not known	pyrethroid pesticide; possible interaction with CYP metabolism
28680–45–7 and 2440-02-0 Heptachloronorbornene (bicyclo[2.2.1]hept-2-ene, heptachloro-)	0	not known	CYP1A (?)
Hexachlorocyclohexane (HCH) isomers	1	yes	endocrine disruptors
107–46–0 HMDS (disiloxane, hexamethyl-)	0	not known	
465–73–6 Isodrin (1,4:5,8-dimethanonaphthalene, 1,2,3,4,10,10-hexachloro-1,4,4a,5,8,8a-hexahydro-, (1.α.,4.α.,4a. β.,5. β.,8. β.,8a. β.)-)	1	yes	
Organic lead compounds	3	no	ALA-D inhibition and chemical analysis
72–43–5 Methoxychlor (benzene,1,1'-(2,2,2-trichloroethylidene)bis(4-methoxy)	1	yes	similar mechanism as DDE (?); eggshell-thinning, endocrine disruptor
Musk xylene	1	not known	range of substances, reproduction, behaviour
Chlorinated naphthalenes	3	not known	CYP1A induction
51000–52–3 Neodecanoic acid, ethenyl ester	0	not probable	
Nonylphenol/ethoxylates (NP/NPEs) and related substances	3	yes	oestrogenic, vitellogenin induction
140–66–9 Octylphenol (phenol, 4-(1,1,3,3,tetramethylbutyl)-)	3	yes	oestrogenic, vitellogenin induction
1825–21–4 Pentachloroanisole	1	not known	
Pentachlorophenol (PCP)	1	not known	pesticide, high acute toxicity
603–35–0 Phosphine, triphenyl-	1	no	oxidative stress
Polycyclic aromatic hydrocarbons (PAHs)	3	no	CYP1A induction, DNA adducts, carcinogenic
Polychlorinated biphenyls (PCBs)	1–3	yes (metabolites)	range of substances with different mechanisms of action
Polychlorinated dibenzodioxins (PCDDs), polychlorinated dibenzofurans (PCDFs)	3	yes	CYP1A induction; embryotoxicity (fish)
Short-chained chlorinated paraffins (SCCP)	0	not probable	membrane stability?
79–94–7 TBBA (phenol, 4,4'-(1-methylethylidene)bis[2,6-dibromo]-)	1	no	immunomodulation (?)
2227–13–6 Tetrasul (benzene, 1,2,4-trichloro-5-[(4-chlorophenyl)thio]-)	0	not known	
87–61–6 Trichlorobenzene (benzene, 1,2,3-trichloro-)	0	not probable	immunomodulation?
120–82–1 1,2,4-trichlorobenzene (benzene, 1,2,4-trichloro-)	0	not probable	immunomodulation?
108–70–3 1,3,5-trichlorobenzene (benzene, 1,3,5-trichloro-)	0	not probable	immunomodulation?
55525–54–7 Urea, N,N'-bis[(5-isocyanato-1,3,3-trimethylcyclohexyl)methyl]-	0	not known	
Cadmium	3		metallothionein induction
Lead	3		ALA-D inhibition
Mercury	3		metallothionein induction
Organic mercury compounds	1	No	Immunomodulation (?)
Organic tin compounds	3	yes	imposex/intersex

Notes: ALA-D: δ-aminolevulinic acid dehydratase; immunomodulation: alteration of the immune system.

7.2.1 Polybrominated diphenylethers (PBDEs)

Updated information is presented on the occurrence of polybrominated diphenylethers (PBDEs) and other brominated flame retardants in the environment and in human tissues. These compounds, which are suspected endocrine disruptors, are widely distributed in the environment and have been found in human samples (e.g., human milk) at levels that give cause for concern. Studies are continuing on the distribution, fate, and effects of these chemicals and temporal trends are being detected in response to changes in production and use patterns.

Recommendations and advice

In line with an earlier ICES recommendation, it was noted that the polybrominated diphenylethers will be included within the OSPAR Joint Assessment and Monitoring Programme (JAMP) as members of the group of brominated flame retardants, and thus will be included in associated monitoring programmes and assessments. ICES recommends that they also be considered for inclusion within the HELCOM COMBINE Programme.

Scientific background

New information on polybrominated diphenylethers (PBDEs) and other brominated flame retardants was presented at the DIOXIN 2002 Conference held in Barcelona. In addition, new information was available from the EU risk assessment on decabromodiphenylether (BDE209, the deca-mix PBDE formulation) and from the European research project FIRE, Risk assessment of brominated flame retardants (BFRs) as suspected endocrine disruptors for human and wildlife health.

In Japan, a 44-fold increase in total BDE concentrations in adipose tissue was observed between 1970 and 2000. The total BDE concentration in human milk samples from Vancouver, Canada was 43 ng g^{-1} lipid weight. This is ten times higher than concentrations observed previously in Sweden, and furthermore had increased by a factor of fifteen during the period from 1992 to 2000. Most blood samples from Sweden contained 1–6 ng g^{-1}, but 5% of the samples had BDE concentrations above 30 ng g^{-1}. High concentrations of BDE209 were found in the blood of rubber industry workers in Sweden. PBDE data for human milk samples were reported from the Czech Republic and Japan, and data were available for PBDEs in blood samples from Korea, and for polybrominated dibenzo-*p*-dioxins (PBDDs) and polybrominated dibenzofurans (PBDFs) in blood samples from Japan. High concentrations of BDE47 (47 ng g^{-1} lipid weight) were found in human serum samples from California.

While food is considered a major exposure route of PBDEs to humans, a German study indicates that household dust may also be an important vector. A Swedish study indicated a three-fold higher human uptake of HBCD (hexabromocyclododecane) from food than was seen for PBDEs.

Investigations of sediment cores from Japan have shown a correlation between the production data for PBDE formulations and the BDE concentrations found in sections of the cores. Extremely high concentrations of PBDEs (up to 2,000 µg kg^{-1} lipid weight) were found in fish samples from Minnesota. Preliminary results from Germany document the occurrence of BDE209 in freshwater fish.

Thirty-six laboratories from thirteen countries took part in a QUASIMEME interlaboratory study on the determination of BDEs, HBCD (hexabromocyclododecane), TBBP-A (tetrabromobisphenol-A), and Me-TBBP-A in fish, mussels, sediment, sediment extract, human milk, and standard solutions of unknown concentrations. The results can be summarized as follows. Results for BDE-7 were satisfactory for all matrices (CV = 15–25%). For BDE99, there was a considerable improvement compared to the first worldwide intercomparison study (CV = 22–35%, exception milk CV = 65%). Analysis of BDE209 was once again not under control in most laboratories. Only a few results were submitted for TBBP-A, Me-TBBP-A, and HBCD. Laboratories need more time to set up methods for the latter three flame retardants before comparable data can be generated.

The situation regarding the environmental distribution of the PBDEs has become complicated due to changes in production and use. Following a risk assessment undertaken within the EU, the use of the penta-mix PBDE formulation has ceased in Europe, and was replaced by enhanced production of two other brominated flame retardant compounds, hexabromocyclododecane (HBCD) and tetrabromobisphenol-A (TBBP-A). As a result, there are now some indications that environmental concentrations of penta-mix PBDEs in Europe are beginning to fall (e.g., in seabird eggs and pike from Sweden). In contrast, North America now has approximately 97% of the world production and use of the penta-mix PBDE formulation, and concentrations in the USA and Canada are continuing to rise rapidly due to the lack of regulation in those countries.

The EU risk assessment of the deca-mix PBDE formulation is due to be completed in June 2003. BDE209 was detected for the first time in peregrine falcon eggs from Sweden in 2001. Samples (eggs, muscle, liver) of peregrine falcons from the UK and of other bird species are currently being analysed to confirm these findings. First results indicate the presence of BDE209 in peregrine falcons from the UK as well as from Sweden.

The EU project FIRE, with nineteen partners from seven European countries, started in 2002. More than 25 brominated flame retardants (BFRs), including BDE28,

BDE47, BDE99, BDE100, BDE153, BDE154, BDE183, BDE209, TBBP-A, α-, β- and γ-HBCD, FR720, 2,4,6-tribromophenol, 6-OH-BDE47, and TB-DEPH (tetrabromophthalic acid, diethylhexyl ester), will be tested using *in vitro* screening assays to select five BFRs for further *in vivo* studies. Human and aquatic wildlife hazards will be identified by *in vivo* studies on rats and on zebrafish and flounder, respectively. Moreover, human exposure and aquatic ecosystem exposure assessment will be performed. Finally, all data will be used for an integrated risk assessment for humans and wildlife.

7.2.2 Phenylurea herbicides (diuron and isoproturon)

Summary

Information is presented on the determination and occurrence in the environment of two phenylurea herbicides, diuron and isoproturon. Initial observations indicate that these herbicides occur widely in the marine environment, with higher concentrations being observed in coastal waters. This distribution may be partly explained by the use of diuron as a booster biocide in antifouling paints, while the distribution of isoproturon is influenced by riverine inputs. The data presented in Annex 6 derive from work undertaken in the North Sea and the Baltic Sea, and demonstrate that the phenylurea herbicides diuron and isoproturon are widely distributed in these sea areas and occur at significant concentrations at some locations.

Recommendations and advice

ICES recommends that Member Countries consider these compounds for inclusion in marine monitoring programmes so as to yield information on their wider distribution and potential for effects.

Scientific background

A review note has been prepared on the presence of two phenylurea herbicides (diuron and isoproturon) in the marine environment. The production volumes, physico-chemical properties, toxicity, analytical techniques, and occurrence of diuron and isoproturon (mainly in the North Sea and Baltic Sea) are included. This review note is contained in Annex 6.

The production data for these compounds need to be updated, but the information is difficult to obtain. For instance, estimates for the EU production of isoproturon in 1995 range from 10–50 kilotonnes per year and German usage is over 1 kilotonne per year.

The phenylurea herbicides are compounds of medium polarity, and because of their relatively high water solubility they are present in the dissolved phase and transported to the marine environment mainly in the water phase. The water phase occurrence and phytotoxic mode of action of the phenylurea herbicides suggest that algae should be one of the primary toxic endpoints for these compounds.

Accordingly, water is the principal matrix in which phenylurea herbicides should be monitored. As these chemicals are thermolabile as well as relatively polar, they are not directly amenable to gas chromatography and so the preferred analytical technique is high-performance liquid chromatography (HPLC) with different detectors. The need for selective and very sensitive detection techniques for the identification and detection of phenylurea herbicides in seawater samples is stressed. The best results have been obtained using an HPLC-MS/MS (tandem mass spectrometry) technique.

Data on the occurrence of phenylurea herbicides in the marine environment are still relatively scarce. Recent results from the German monitoring programme of 2000 to 2002 demonstrate the large geographical extent of contamination of the coastal and offshore waters by phenylurea herbicides in the Elbe estuary, German Bight, Baltic Sea, and North Sea, and also in the English Channel. Isoproturon and diuron were detected more frequently and at higher concentrations than the other compounds determined, but other herbicides (chlortoluron, fenuron, linuron, and monolinuron) were also detected. The geographical distributions in the German Bight show pronounced gradients for all herbicides, with declining concentrations with increasing distance offshore linked to mixing and dilution processes. The occurrence of diuron and isoproturon in the waters of the North Sea and Baltic Sea is distinct and probably reflects the different uses of these two compounds, at least in part. The contamination of coastal waters by diuron close to harbours could be related to its use as a booster biocide in antifouling paints, whilst the geographical distribution of isoproturon is probably more strongly influenced by riverine inputs.

The reported concentration levels of the dissolved phenylurea herbicides in the North Sea and Baltic Sea, which are within the range of < 1 ng l^{-1} to > 10 ng l^{-1}, are higher than those of organochlorine insecticides, such as the hexachlorocyclohexane (HCH) isomers, at the same sampling stations.

Diuron and isoproturon are included in the list of priority pollutants given in the Water Framework Directive (WFD) and should be considered as candidate contaminants for future monitoring programmes in the marine environment.

8 FISH DISEASE ISSUES

8.1 Trends in Diseases of Wild and Farmed Fish and Shellfish

Request

This is part of continuing ICES work to consider information on new developments with regard to fish and shellfish diseases that ACME disseminates to ICES Member Countries in order to inform them of present and potential future problems.

Source of the information presented

The 2003 report of the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) and ACME deliberations.

Summary

This section contains a compilation of the most recent information on recent outbreaks and new disease trends in wild and farmed fish and shellfish (molluscs and crustaceans) submitted by Member Countries.

Information is provided on viral and bacterial diseases as well as on diseases caused by fungi, parasites, and other factors. New findings considered to be of particular relevance are:

- the first report on the occurrence of Infectious Salmon Anaemia Virus (ISAV) in rainbow trout in Ireland;
- the first report on the occurrence of *Parvicapsula* sp. in salmon farms in Norway;
- the first report on the occurrence of *Haplosporidium nelsoni* and *H. costale* in the oyster *Crassostrea virginica* in Canada;
- the increasing number of reports on diseases in commercially important crustacean species.

Recommendations and advice

ICES recommends that Member Countries ensure that adequate funding is made available to continue health surveillance of wild fish and shellfish stocks, as continued disease monitoring is necessary to:

a) be used as an indicator of environmental conditions;
b) assess the impact of disease in wild stocks;
c) assess the potential for disease interactions between wild and farmed fish and shellfish;
d) recognize emerging diseases caused by infectious agents and/or contaminants;
e) understand the mechanisms of disease transmission and epizootic events.

Based on the review of new developments regarding diseases of wild and farmed fish and shellfish, ICES further recommends that Member Countries conduct specific studies on the following issues:

1) evaluation of the impact and risk of Aquareovirus for farmed fish;
2) identification of whether wild fish are reservoirs of *Parvicapsula* sp. and determination, by undertaking transmission trials, of whether horizontal transmission is possible between salmon;
3) evaluation of the impact of jellyfish on farmed fish;
4) determination of whether the finding of the protozoan parasites *Haplosporidium nelsoni* and *Haplosporidium costale* in the Cape Breton area, Nova Scotia, Canada, represents new introductions or whether the parasites were present earlier and, if the former, how the introduction occurred;
5) evaluation of the occurrence of viral diseases in crustaceans, including the herpes-like virus in *Panulirus argus*, to specifically identify agents and their effects on various crustacean species;
6) identification of the haplosporidian infecting *Carcinus maenas* using molecular techniques and determination of its effects on crab populations.

Scientific background

The ACME reviewed the section of the WGPDMO report on new disease trends reported by the country representatives. Special attention was drawn to the distribution and prevalence of the diseases in wild and farmed fish and shellfish listed below.

Wild fish

Viral diseases:

Viral haemorrhagic septicaemia virus (VHSV) continues to increase its host spectrum, distribution, and occurrence. In the U.S., ten new isolates of the North American strain were isolated from various species of fishes in California and Washington. Nine of these isolates may represent a genotypic subgroup distinct from North American strains previously isolated from Washington State, Alaska, and British Columbia. The European strain of VHSV has been isolated from sandeel (*Ammodytes* sp.) (representing a new host species for the European strain of the virus), and genotype I of VHSV was isolated from a wild Atlantic salmon (*Salmo salar*) in Spain diseased with *Aeromonas salmonicida*.

While the overall prevalence of lymphocystis in North Sea dab (*Limanda limanda*) showed no new trends, slight increases were found in dab at two locations in the Irish Sea and at the Flamborough Ground and Dogger Bank.

The prevalence continues to decrease in the German Bight. Higher prevalences were reported in Baltic flounder (*Platichthys flesus*), with maximum levels in the southwestern Baltic off the Polish coast.

Bacterial diseases:

Two new mycobacterial species (*Mycobacterium shottsii* and *M. chesapeaki*) have been identified from striped bass (*Morone saxatilis*) in the Chesapeake Bay, USA. *M. shottsii* is the predominant organism in the on-going epizootic and was found in 76% of striped bass with and without external signs of disease.

An increased prevalence of acute/healing skin ulcerations was found in dab from a few locations in the Irish Sea and in Baltic cod (*Gadus morhua*) near Gdansk Bay, but the prevalence has decreased in cod from the southwestern Baltic.

Algal kills:

The first fish kill (menhaden *Brevoortia tyrannus*) due to *Karlodinium micrum* was reported in Chesapeake Bay, USA, in 2002.

Parasites:

Gyrodactylus salaris remains a major threat to Atlantic salmon in Norway and two additional rivers became infested in 2002. *Pseudodactylogyrus* sp. was found for the first time in the gills of eel (*Anguilla anguilla*) collected from the southeastern Baltic Sea.

The sea louse *Lepeophtheirus salmonis* is implicated in unusually low returns of pink salmon (*Oncorhynchus gorbuscha*) in western Canada, and continues to be a major problem for wild salmon and sea trout (*Salmo trutta*) in Norway. An increasing trend in the prevalence of sea lice has been recorded in wild salmon returning to rivers in Maine, USA, in the period from 1998 to 2002.

Other diseases:

There has been a significant decline in the prevalence of liver nodules (macroscopic neoplastic and pre-neoplastic lesions) in North Sea dab at the historical hot spots, suggesting that prevalences are approaching natural background levels.

Farmed fish

Viral diseases:

In July 2002, Infectious Salmon Anaemia Virus (ISAV) was isolated from rainbow trout (*Oncorhynchus mykiss*) at two isolated marine sites in the west of Ireland. All other farms in the country have been tested for ISA (using virus isolation), and found to be negative.

Aquareovirus was isolated from haddock (*Melanogrammus aeglefinus*) and Atlantic halibut (*Hippoglossus hippoglossus*) in eastern Canada. In some cases, infections were associated with liver pathology. In Scotland, moribund Atlantic halibut larvae were sampled for a suspect nodavirus infection in August 2002. At present, the aetiology is a suspect Aquareovirus.

Bacterial diseases:

Edwardsiella tarda causing pathology and mortality (10%) was reported from a turbot (*Scophthalmus maximus*) farm in France and represents the first isolation since 1991.

Parasites:

Clinical disease outbreaks caused by *Parvicapsula* sp. occurred in Atlantic salmon in Norway. This myxosporean was diagnosed in five salmon farms in northern Norway. A large number of parasites were found especially in the pseudobranch, but also in the liver and kidney. The mortality ranged from 3% to 50%. This is the first known outbreak of disease caused by this parasite in Norwegian farmed fish.

Other diseases or events:

In Scotland, twenty farms rearing Atlantic salmon reported losses due to jellyfish swarming against the cages, with a total loss of 1,815,800 fish. Several fish farms in Norway experienced losses due to jellyfish (*Muggiaea atlantica*). The reason for this may be the exceptional, high water temperatures in late summer and early autumn.

Wild and farmed shellfish

Viral diseases:

A novel intranuclear bacilliform virus (IBV) (Baculo-like virus) was reported in wild-caught brown shrimp (*Crangon crangon*) from UK estuaries. The virus causes necrosis of the hepatopancreas and midgut epithelial cells. A prevalence of up to 100% was found at sites in the Clyde estuary and high levels (up to 85%) were found at sites in the Forth and Mersey estuaries. Other sites are currently being assessed.

A herpes-like virus (HLV) has been discovered in wild populations of the Caribbean spiny lobster (*Panulirus argus*) and appears to be the first reported viral disease of lobsters. The HLV-PA occurs in as many as 16% (N = 850) of the juvenile spiny lobster populations sampled from the mid- to lower Florida Keys in the Gulf of Mexico, USA. The virus attacks haemocytes causing the blood to become milky white, and appears to affect juveniles only. The virus is highly lethal in experimental conditions.

Fungi:

The appearance of an unidentified organism, interpreted as a yeast, was reported in the cockle, *Cerastoderma edule,* in the Croisic area (Loire Atlantique, France) after the "Erika" oil spill. The presence of the "yeast" was associated with a massive cellular reaction. A causative role of the oil pollution in the appearance of the infection is suspected, but has not been demonstrated.

An unidentified yeast was found in virus-infected wild brown shrimp from UK estuaries.

Parasites:

Bonamia ostreae was detected for the first time since 1994 (when one of 450 oysters was found to be *Bonamia*-positive) in a bed of wild oysters in Achill Sound, Co. Mayo, Ireland. The prevalence of infection was 13% (of 150 oysters) in this bed, but remained unchanged in other affected zones in Ireland (< 20%). Achill Sound thereby lost its EU-approved zone status (2002/300/EC) with respect to *B. ostreae*.

Haplosporidium nelsoni (MSX) and *H. costale* (SSO) were reported for the first time in oysters (*Crassostrea virginica*) from Canada (Cape Breton, Nova Scotia). *H. nelsoni* infections caused heavy, although localized, mortalities (> 90%) on several leases in the Bras d'Or Lakes, Cape Breton, Nova Scotia. To date, infections appear to have a tight distribution focus related to the St. Patricks Channel area of Bras d'Or Lakes. *H. costale* was found at low prevalence (< 5%) on the north shore of Cape Breton during the intense surveillance effort mounted to map the distribution of MSX. Several samples that showed "light background" plasmodial infections but no indication of pathology were found on the north shore of Cape Breton, as well as at several sites within the Gulf of St. Lawrence.

A novel infection with *Haplosporidium* sp. was detected in approximately 10% of shore crabs (*Carcinus maenas*) captured from a southern England site. The parasite causes lethargy and loss of aggressiveness in infected crabs. The haemolymph appears opaque and creamy and coats internal organs.

The protozoan parasite *Mikrocytos mackini* was reported for the first time in the USA in Washington State in May 2002. The organism was detected in two feral populations of *Crassostrea gigas* at a prevalence of 6% (mild infections) in each population, as determined by both histology and polymerized chain reaction (PCR). More than fifteen other populations of wild and farmed broodstock Pacific oysters were then surveyed throughout Washington with no positive results. The disease agent requires temperatures below about 15 °C and closer to 10 °C to manifest itself, which may limit its southern distribution. There are two incident reports of a similar organism occurring in the native Olympia oyster (*Ostrea conchaphila*), which is known to be susceptible to the disease. The disease agent has probably not moved southward. More likely, it has been present in feral old oysters at very low prevalence and intensity for some time and was never detected because such oysters were never examined.

Other diseases:

A new disease syndrome, described as an excretory calcinosis, was observed in lobster (*Homarus americanus*) in Long Island Sound, USA, during the summer of 2002, when sustained elevated temperatures and hypoxic conditions were associated with lobster mortality.

In Germany, a study on histopathological disorders in wild blue mussel (*Mytilus edulis*) populations during the period 1994–2000 revealed an elevated prevalence of haemocyte infiltration, granulocytomas, and pathological disorders of the gonads (atrophy and atresia), possibly linked to higher levels of tributyltin (TBT) and triphenyltin (TPT) in animals in harbour areas.

8.2 Status of the M74 Syndrome in Baltic Salmon and Status of *Ichthyophonus* in Herring

Request

This is part of continuing ICES work to update knowledge on the causes of the M74 syndrome in Baltic salmon and progress in understanding the implications of relevant environmental factors, and on the status of *Ichthyophonus hoferi* infection in herring.

Source of the information presented

The 2003 report of the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) and ACME deliberations.

Summary

This section includes an update on knowledge of the status of the M74 syndrome in Baltic salmon (*Salmo salar*) and on progress in understanding the underlying causes. It further provides information on the status of the infection of herring (*Clupea harengus*) and other fish species with *Ichthyophonus hoferi*.

From current data, there is alarming evidence that the number of female Baltic salmon producing offspring affected by the M74 syndrome is increasing. This finding indicates that the situation of the wild stock of Baltic salmon is unlikely to improve. No significant new information is available on the causes of the M74 syndrome.

Ichthyophonus hoferi infection persists at low prevalence in Atlantic herring stocks from the northern North Sea and the Barents Sea. The high prevalence reported in Pacific herring is discussed as a possible cause of the decline in the adult herring population.

Recommendations and advice

Owing to the continuous significance of the M74 syndrome in Baltic salmon, ICES urges relevant Member Countries to provide sufficient resources for continued studies into the aetiology of M74, specifically addressing the role of food organisms and contaminants.

ICES recommends that Member Countries continue to monitor the prevalence of *Ichthyophonus hoferi* infection in herring stocks, if appropriate in connection with stock assessment surveys, so as to be aware of changes in status that may forecast an epizootic disease outbreak.

Scientific background

M74

The ACME noted with concern that, as predicted in 2001, current data strongly indicate that the prevalence of adult female salmon producing offspring affected by the M74 symptoms has continued to increase in most of the Baltic rivers compared to the two preceding years. No reliable data are available at present for a prognosis for the 2003 hatch.

Despite the urgent need to identify the causes of the syndrome, still no significant breakthrough has been reported in the research on the aetiology of the M74 syndrome. Most studies still focus on thiamine deficiency maternally transferred to the offspring or on thiaminase activity in the food chain of Baltic salmon. The data thus far suggest that the immediate cause behind the thiamine deficiency syndrome of salmon is the high thiaminase activity of the salmon prey species, Baltic herring, but not of sprat (*Sprattus sprattus*). No evidence was found in support of the previous hypothesis that algal blooms are a main factor in the aetiology of the M74 syndrome. The role of planar PCBs and similar compounds in the aetiology of M74 is still in focus. The concentrations of these compounds were seen to increase coincidentally with the outbreak of M74, but a direct link remains to be demonstrated.

***Ichthyophonus hoferi* infection**

The fungal parasite *Ichthyophonus hoferi* was isolated from four new fish hosts (Pacific tomcod (*Microgadus proximus*), speckled sanddab (*Cihtarichthys stigmaeus*), Puget Sound rockfish (*Sebastes emphaeus*), and surf smelt (*Hypomesus pretiosus pretiosus*)) in Puget Sound, USA, and from Atlantic salmon from Russian rivers flowing into the Barents Sea. High prevalences reported in Pacific herring (*Clupea pallasii*) are suggested to be associated with the unexplained decline in the adult herring population in the Pacific. *I. hoferi* infections continue to persist at low prevalence in Atlantic herring from the northern North Sea and the Barents Sea.

8.3 Strategies to Assess the Prevalence of Shellfish Diseases in Parallel to Fish Diseases and Chemical Contaminant Levels in Environmental Monitoring Programmes

Request

This is part of continuing ICES work to consider new developments with regard to the monitoring of fish and shellfish diseases, and the use in monitoring the effects of contaminants.

Source of the information presented

The 2003 report of the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) and ACME deliberations.

Summary

Fish diseases have long been used by several ICES Member Countries as part of a suite of techniques employed to determine biological effects of contaminants, but only a few studies have incorporated molluscan or crustacean pathology or disease in combination with fish disease and contaminant data. An example of where this has been adopted is the multi-year National Status and Trends Program that covered a large number of sites on all coasts of the USA and included the assessment of fish and shellfish species, using an integrated sampling approach including contaminant levels. It is now continuing with shellfish and contaminants. No such integrated programmes currently exist in other ICES Member Countries, although there are research projects which incorporate both fish and shellfish diseases in assessments.

It is emphasized that clear strategic objectives for the monitoring are crucial in deciding whether to include several target species, including shellfish. For assessment of the environmental impact of pollution incidents such as oil spills, it is desirable to examine the disease status of a broad range of species appropriate for the region concerned and these would include shellfish species. For longer-term monitoring, sentinel species for detecting the impact of contaminants on health should be used, this being the current strategy employed by a number of national monitoring programmes. However, these do not routinely utilize shellfish species. It is recognized that there are limitations concerning the proposed use of shellfish as target organisms for monitoring. These include the lack of baseline data on normal tissue and organ structure, immunological parameters, and host responses to contaminants. However, the use of histopathology to detect host tissue responses is thought to be valuable in providing information on the overall health status of target shellfish species because more specific tools will not be available. Regarding the observation of parasite pathogens and associated host responses, it is considered that these may provide important data on individual and population health status. The use of such data for the assessment of contaminant effects is more problematic unless the life cycles, including requirements of intermediate hosts of the parasites, are known.

It is considered possible for biological effects monitoring to use material collected during national monitoring programmes for EU-listed notifiable diseases (e.g., Bonamiosis and Marteiliosis) and for the assessment of algal toxins to include contaminant effects. Since the diagnosis of notifiable diseases is mainly based upon the examination of histological sections, these can readily be assessed for the provision of additional data on more general pathological effects as well as disease intensity and incidence, which could be used in the context of environmental assessment.

Recommendations and advice

ICES recommends that Member Countries increase integration of monitoring efforts regarding diseases and biological effects of contaminants in fish and shellfish in coastal and offshore marine environments in order to maximize the data obtained from samples taken and to avoid duplication in sampling effort. Data generated as part of studies on algal toxins in molluscs should be utilized as appropriate.

Scientific background

Diseases of fish and shellfish may be used as indicators of marine pollution. The WGPDMO has developed standardized methodologies for monitoring marine fish external diseases and flatfish liver pathology. These methods are used by ICES Member Countries to investigate the relationship between changes in the marine ecosystem and the prevalence of fish diseases. In parallel, standardized methodologies have been developed to evaluate the prevalence of imposex/intersex in marine gastropods as an indicator of TBT contamination. Currently, within most monitoring programmes, surveys of fish and shellfish diseases are not integrated: they are conducted at different times and sites, and have different objectives. The objective here is to evaluate the potential usefulness of integrating the monitoring of fish and shellfish diseases to evaluate marine ecosystem health.

Advantages and disadvantages of using fish or shellfish for monitoring the effects of contaminants on the health of marine organisms

Several criteria are used to select sentinel species for marine monitoring programmes. The species should be abundant and widespread, easy to sample, non-migratory and preferably not very mobile, and sediment-dwelling. They should have a high potential for exposure and a measurable response to contaminants. Species with a higher fat content, a longer lifespan, and located at a high trophic level are more likely to accumulate persistent pollutants. Filter feeders, such as bivalves, may also accumulate high concentrations of contaminants in their tissues. Several fish and shellfish species fulfil these criteria. Generally, no single species is an ideal sentinel for all contaminant groups and vulnerability varies markedly among groups of contaminants.

Fish have stronger metabolic capacities (e.g., cytochrome P4501A1) than shellfish. Thus, compared to shellfish, they may be more sensitive to contaminants that are bioactivated by the cytochrome P450 system (such as carcinogenic PAHs) but may be less sensitive to toxicants that are detoxified by cytochrome P450. Toxicology, pathology, physiology, and endocrinology have been studied more intensely in fish than in shellfish. Cause-effect relationships between exposure to contaminants and histological changes have been more extensively documented in fish than in shellfish. In the 1970s, most environmental programmes focused on persistent organic pollutants. Top-predator fish were useful sentinel species for these studies. In the past ten years, the focus has been on new classes of contaminants, including endocrine-disrupting compounds (EDCs), pesticides, and pharmaceutical compounds, many of which do not accumulate in fish tissues and are more concentrated in the coastal areas.

Because they are generally sedentary, shellfish species may be particularly useful to localize point sources of contamination and to evaluate their effects. Cage studies may be easier with shellfish (particularly with bivalves) than with fish and are very useful for conducting experiments in the field to demonstrate cause-effect relationships. Invertebrates, particularly the psodobranch snails, are sensitive to endocrine disruption at environmentally relevant concentrations. In freshwater and marine (*Nucella lapillus*, *Nassarius reticulatus*) psodobranch species exposed in the laboratory, xeno-estrogens (e.g., bisphenol A, octylphenol) caused induction of superfemales and reduction in the size of the male sex organs. Xeno-androgens (triphenyltin, tributyltin) caused virilization of females (imposex). Anti-androgens (cyproterone acetate, vinclozolin) caused reduced male sexual organs and suppression of imposex development (Oehlmann *et al.*, 2000; Schulte-Oehlmann *et al.*, 2000; Tillmann *et al.*, 2001).

The relative simplicity of invertebrate immune functions offers a good model to study the complex interactions between exposure to environmental contaminants and immune dysfunction (Galloway and Depledge, 2001). In 2001, IFREMER (Institut Français de Recherche pour l'Exploitation de la Mer) funded a multi-year study called MOREST (2001–2005) to investigate summer mortality in Pacific oysters (*Crassostrea gigas*) that may be caused by multiple factors including elevated temperatures, the physiological stress associated with maturation, aquaculture practices, pathogens, and contaminants. Recent studies have shown that oyster defenses may be influenced by contaminants (Fournier *et al.*, 2002; Sauvé *et al.*, 2002). However, possible relations between contaminants, the physiological state of oysters, their immune system, and pathogens need further investigation. In France, Gagnaire *et al.* (2003) studied haemocyte responses (ratio between granulocytes and hyalinocytes, phagocytosis, percentages of cells possessing hydrolytic enzymes and reactive oxygen species production) *in vitro*, *in vivo*, and *in situ* in the presence of contaminants (heavy metals, pesticides, herbicides such as atrazine).

Several fish and shellfish diseases, with the potential to cause deleterious impacts on fish and shellfish populations, have been associated with exposure to environmental contaminants. Most of these conditions have a multifactorial aetiology and may be triggered by a variety of natural and anthropogenic factors. Imposex in gastropods is used successfully as a biomarker of exposure to tributyltin in the marine environment in several ICES Member Countries. Haemic and gonadal neoplasia in bivalves has been associated with environmental contamination by herbicides, but a cause-effect relationship has not yet been fully demonstrated. Hepatic pre-neoplastic and neoplastic lesions in fish are used successfully as a biomarker of exposure to environmental carcinogens, particularly PAHs. Prevalence of intersex in fish is used as an indicator of contamination of the marine environment by oestrogenic compounds. Thus, histopathological lesions in fish and shellfish are presently used to detect different groups of contaminants and are complementary.

Review of case studies where the prevalence of diseases was assessed in both fish and shellfish species as an indicator of environmental contamination

While numerous studies have been published on the association between fish diseases or shellfish diseases and environmental contamination, very few studies have looked simultaneously at the prevalence of fish and shellfish diseases at contaminated sites.

Carcinogens:

The eastern oyster (*Crassostrea virginica*) developed neoplastic disorders when experimentally exposed both in the laboratory and in the field to chemically contaminated sediment from Black Rock Harbor (BRH), Bridgeport, Connecticut, USA. High concentrations of polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), and chlorinated pesticides were measured in BRH sediment and in oysters exposed to the sediment in the laboratory and in the field. Neoplasia was observed in oysters after 30 days and 60 days of continuous exposure in a laboratory flow-through system to a 20 mg l^{-1} suspension of BRH sediment plus post-exposure periods of 0, 30, or 60 days. Tumour occurrence was highest in the renal excretory epithelium, followed in order by gill, gonad, gastrointestinal, heart, and embryonic neural tissue. In field experiments, gill neoplasms developed in oysters deployed in cages for 30 days at BRH. The uptake of PAHs and PCBs from BRH sediment observed in oysters also occurs in blue mussels (*Mytilus edulis*). Winter flounder (*Pseudopleuronectes americanus*) fed BRH-contaminated blue mussels contained the same xenobiotic chemicals as analysed in mussels. The flounder developed renal and pancreatic neoplasms and pre-neoplastic hepatic lesions, demonstrating trophic transfer of sediment-bound carcinogens up the food chain (Gardner and Yevich, 1988; Gardner *et al.*, 1991).

Petroleum hydrocarbons:

Field studies conducted following the "Amoco Cadiz" oil spill have shown pathological changes in several shellfish and fish species. In April–May 1978, cockles (*Cerastoderma edule*) and mussels (*Mytilus edulis*) were deployed in cages for 25 days in the Baie de Morlaix, a site heavily oiled by the "Amoco Cadiz", and in the Rade de Brest, a control site. Mussels and cockles from the contaminated site had a much higher accumulation of lipid and lysosomal granules in their digestive diverticula compared to mussels from the control site. Plaice (*Pleuronectes platessa*) were sampled in two estuaries heavily oiled by the "Amoco Cadiz", the Aber Wrac'h and the Aber Benoit, and in reference sites located on the western and southern coasts of Brittany. Several histopathological changes were found at oil-contaminated sites, including fin and tail necrosis, hyperplasia and hypertrophy of gill lamellar mucous cells, increased density of pigmented macrophage aggregates in the liver, and delayed maturation of the oocytes (Haensly *et al.*, 1982; Stott *et al.*, 1983).

In both of these studies, fish and shellfish were exposed to complex mixtures of contaminants and developed different types of histopathological changes. It is not known whether they responded to the same group or to different groups of chemicals. The demonstration of toxic-induced pathological changes in both fish and shellfish species has contributed to assessing the ecological impact of the contaminants and has suggested the possibility of trophic transfer of contaminants.

In a study on biomarkers of pollution stress in marine molluscs and fish in littoral regions in the Red Sea, Mediterranean Sea, and the North Sea, researchers found an elevated frequency of DNA lesions (alkaline and acidic DNA unwinding), increased prevalence of micronucleus-containing haemocytes, and organ pathologies in samples from contaminated sites (Bresler *et al.*, 1999).

Review of existing environmental monitoring programmes assessing the prevalence of shellfish diseases in parallel to fish diseases (at the same sites and times)

Few environmental monitoring programmes integrate the evaluation of fish and shellfish pathological and histopathological changes.

In Canada, studies have been carried out on histopathology in mummichogs (*Fundulus heteroclitus*) at the same sites that were used for studying immunological biomarkers and the incidence of haemic neoplasia in caged mussels. Fish and mussels were collected in the areas within and around Pictou Harbour, Nova Scotia, which receive the effluent of a pulp and paper mill. Analyses of the results and preparation of a publication are under way.

In the UK, the research programme "Endocrine Disruption in the Marine Environment" (EDMAR) (1998–2001) (Allen *et al*., 2002) investigated the effects of endocrine disruptors (ED) in three fish species, European flounder (*Platichthys flesus*), viviparous blenny (*Zoarces viviparus*), and the sand goby (*Pomatoschistus minutus*), and in one crustacean species (*Cancer pagurus*) from estuarine environments from around the UK. Flounder and blenny were found to exhibit increased vitellogenin (VTG) induction and the presence of intersex in estuaries contaminated with endocrine-disrupting chemicals. Sand gobies were not found to be susceptible to either elevated VTG levels or intersex induction. However, morphologically intermediate papilla syndrome (MIPS) (Kirby *et al*., 2002) was described in gobies from several locations. Increased induction of vitellin (crustacean analogue of VTG) was not found to occur in crabs exposed to EDs either naturally or experimentally. Firm evidence of changes to the gonads and secondary sexual characteristics was also lacking. Histological evaluation of gonad tissue and other organs was not undertaken. Brown shrimp (*Crangon crangon*) sampled in the same estuaries were also examined but did not show evidence of endocrine disruption.

In a current UK DEFRA-funded programme, investigations based at CEFAS Weymouth Laboratory are evaluating biological effects of contaminants in estuarine organisms. The main aim of this work is to provide a more holistic assessment of effects of contaminants in estuarine environments by integrating biological effects measurements from several organisms. The study includes the same fish species as were targeted in the EDMAR programme, but extends this to incorporate other species found to be abundant at particular locations. In addition, crustacean species are also sampled at the same time to provide an insight into the pathological responses exhibited by resident shellfish. Molluscs are also under investigation but samples are difficult to obtain using the small trawls used to collect fish and crustacean samples.

The study utilized external disease (following ICES guidelines) and multi-organ histopathological biomarkers in all species sampled. In addition, concurrent samples were analysed for gene mutation analysis, DNA adduct formation, and micronuclei analysis (in fish). Data on parasite prevalence and abundance are used to augment the assessment of individual and population health and appear to be a powerful tool to discriminate between impacted and reference sites.

Results to date indicate that liver histopathology is effective in identifying endpoint effects of contaminants in estuarine environments (Stentiford *et al*., 2003) and intersex provides a good endpoint marker for ED exposure. Preliminary data on crustacean pathology have identified several types of lesions associated with infectious agents that may be of use for health assessment in this group. Analysis of mollusc samples and the extended observation of fish and crustacean samples are ongoing.

In the years 2001–2002, a practical ICES Workshop on Biological Effects of Contaminants in Pelagic Ecosystems (BECPELAG) was held to evaluate biological effects of contaminants in pelagic ecosystems: a coastal area (German Bight) and an area in the vicinity of oil rigs (Statfjord). In addition to assessing biological effects in wild-caught organisms, pathological changes and a suite of biomarkers were evaluated in blue mussels (*Mytilus edulis*) and in Atlantic cod (*Gadus morhua*) exposed in cages at each site (see Section 4.1.1, above, and http://www.niva.no/pelagic/web).

In the EU-funded project Biological Effects of Environmental Pollution in Marine Coastal Ecosystems (BEEP) (2001–2004) (http://beep.lptc.u-bordeaux.fr/_summary.asp), biomarkers are assessed in fish and mussels from the Baltic Sea, Mediterranean Sea, and the North Atlantic and are used as indices of stress and of alteration of organism performance. One objective of the project is to identify new sensitive species and species with different life strategies in the ecosystem that can be monitored. Multiple biomarker suites will be developed to be applied in different biota (including both fish and shellfish species) with different environmental toxins.

Conclusions

Pathological changes in fish and shellfish species may be used to indicate the presence and effects of toxic contaminants in the marine environment. The combined use of fish and shellfish species in monitoring studies is advantageous since fish and shellfish species differ in terms of their vulnerability to various groups of contaminants. Very few studies have investigated pathological responses of fish and shellfish species collected at the same study sites. However, in these studies, a variety of histopathological changes were observed in fish and shellfish species exposed to complex mixtures of contaminants and it is not known whether they responded to the same group or different groups of chemicals. Recent interest in the effect of endocrine-disrupting compounds on marine biota has triggered research into the comparative toxicopathology of various fish and shellfish species. These studies will contribute to identifying additional sensitive species and histopathological biomarkers.

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8.4 Impact of Diseases of Farmed Fish on Wild Fish 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 2003 report of the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) and ACME deliberations.

Summary

This section provides information on disease and parasite interactions between wild and farmed fish.

The aquaculture industry is expanding into new fish species, resulting in more complex dynamics for disease interaction and greater concern for the health of wild fish stocks. Therefore, it is essential that ICES provide advice based on the most recent knowledge of this problem.

Most bacterial and viral fish diseases have a broad host range and there is potential for the transfer of infection from wild to farmed fish, and similarly from farmed to wild fish. Viral, bacterial, and parasitic diseases represent a significant threat to aquaculture animals, but in the wild, the impacts are largely unknown. Prevalence, incidence, and the pathology of diseases with the potential to be transmitted between farmed and wild fish have not been widely investigated in wild stocks and there are practical difficulties involved in measuring them.

The ACME reviewed examples of diseases and parasites that have the potential to be transferred from farmed to wild fish and vice versa (Infectious Pancreatic Necrosis (IPN), diseases caused by Nodaviridae, Infectious Salmon Anaemia (ISA), Infectious Haematopoietic Necrosis (IHN), Viral Haemorrhagic Septicaemia (VHS), diseases caused by Aquareovirus, Sleeping Disease (SP), Salmon Pancreas Disease (SPD), diseases caused by bacterial infections, e.g., furunculosis and vibriosis; diseases caused by parasites, e.g., sea lice, *Lepeophtheirus salmonis*, and *Gyrodactylus salaris*).

Recommendations and advice

Because the aquaculture industry is expanding into new fish species, resulting in more complex dynamics for disease interaction and greater concern for the health of wild fish stocks, ICES recommends that relevant Member Countries conduct further studies to evaluate the risk of disease transmission from farmed fish to wild fish, and equally the risk from wild fish infecting farmed stock, by applying appropriate epidemiological methods.

It is further recommended that:

1) emerging viral diseases be assessed for the risk to aquaculture animals and the risk to wild fish;
2) the significance of parasitic diseases (e.g., *Gyrodactylus salaris*) on hatchery production be assessed;
3) further studies be conducted to assess the impact of sea lice and furunculosis on wild salmonid fish stocks.

Scientific background

Atlantic salmon (*Salmo salar*) and rainbow trout (*Oncorhynchus mykiss*) are traditional species for aquaculture, but the decline in catches of wild Atlantic cod (*Gadus morhua*) has led to an increase in the market value and stimulated interest in farming of this and other species. For example, Atlantic halibut (*Hippoglossus hippoglossus*) has been farmed for several years in Norway, the UK, Iceland, and Canada and annual production is increasing. Haddock (*Melanogrammus aeglefinus*) and Dover sole (*Microstomus pacificus*) are likely to be farmed in increasing numbers in the near future. Several RNA and DNA viruses infecting fish have been studied intensively and many induce diseases resulting in economic losses especially in aquaculture. The significance of several important RNA viruses such as orthomyxo-, paramyxo-, rhabdo-, toga-, picorna-, noda- and birnaviruses, and DNA viruses such as parvo-, irido-, and adeno-viruses has been reviewed by Essbauer and Ahne (2001). There are examples of carrier status and equally examples where mass mortality has been attributed to virus infections. Clearly any increase in the diversity of fish species increases the opportunity for disease interaction, however, there are practical difficulties in measuring the prevalence, incidence, and the pathogenicity of diseases in wild stock and the factors associated with the host and environment have not been widely investigated. As most bacterial, parasitic, and viral diseases have a broad host range, there is also the potential for transfer of infection from wild to farmed fish, and equally from farmed to wild fish. It should be emphasized that knowledge of disease transfer and the impact of disease on wild fish is limited.

Viral infections

Infectious pancreatic necrosis virus (IPNV):

Infectious pancreatic necrosis (IPN) is a highly contagious viral disease of young fish, particularly of salmonid species held under intensive rearing conditions. IPN virus (IPNV), or viruses showing serological relatedness to IPNV, have been reported to cause disease in farmed marine fish species, including yellowtail (*Seriola quinqueradiata*), turbot (*Scophthalmus maximus*), halibut (Wood *et al*., 1996), Atlantic salmon (Sousa *et al.,* 1996), sea trout (*Salmo trutta*), and Atlantic cod (*Gadus morhua*) (Rimstad *et al*., 1990; Crane *et al*., 2000). The first report of IPNV in farmed Atlantic cod occurred in fry on the west coast of Denmark during 1991, with almost total loss of the stock. Between 1992 and 1994, Lorenzen *et al.* (1995) also reported a similar high mortality among farmed cod fry from the Faroe Islands. Likewise, IPNV has been detected in wild caught chinook salmon (*Oncorhynchus tshawytscha*) in New Zealand after an estimated 2–3 years in marine waters (Tisdall and Phipps, 1987) and in wild trout from Nova Scotia, Canada. The virus is widespread in farmed fish, particularly in Norway and Scotland, and an increasing number of species are found carrying the virus. Hatcheries and smolt mortality particularly in the Shetland Isles, Scotland is increasing; for example, in Shetland the prevalence of IPNV among farmed fish in marine waters increased from 44% to 75–80% in 1999 and 2000. In Shetland's freshwater sites, prevalence rose from 16% to 30–40%. Clinical IPN in Scotland has also increased from a prevalence of 1.1% to 12.5% for the whole of Scotland from 1988–2002. The source of the virus is speculative, and could include the transfer of infected fish from fresh water, failure of fish to adapt to sea water, strain differences emerging in certain areas, and expression of low-level virus or a wild reservoir (Bruno and Woo, 2002). IPNV is stable for long periods in sea and brackish waters and this could also have an impact on seawater outbreaks of IPNV and transmission from wild to farmed stock, and vice versa.

Wild fish surveys carried out in Scotland to determine the prevalence of IPNV in wild marine and freshwater fish are detailed below.

1) A survey of Atlantic salmon and brown trout (*Salmo trutta (morpha fario)*) captured by electrofishing at eleven freshwater river sites from eleven areas in Scotland was carried out. Juvenile salmon were taken in preference to brown trout and in total 330 fish were examined, comprising 222 Atlantic salmon parr and 118 brown trout. Two out of three pools from salmon were positive for IPNV from the River Arnisdale. This site had been stocked with salmon fingerlings obtained from a supplier specializing in producing salmon and sea trout from wild origin. This survey area was also located near a large farm site that had also tested positive for IPNV.
2) A demersal trawl survey was conducted at locations in the North Sea, namely offshore from the Firth of Forth, Moray Firth and Fair Isle, where a total 2,753 fish consisting of fifteen species were sampled. One haddock was virus positive and additional work is required to identify the isolate. A further ten locations were sampled on the west coast of Scotland in the Kyle of Lochalsh area comprising 3,765 fish (eighteen species). All these samples were tested by tissue culture and were virus negative.
3) A survey in the Firth of Clyde comprised 779 fish (fourteen species). One non-IPN virus positive sample is currently being identified.
4) Freshwater fish were obtained from fifteen locations throughout Scotland using electrofishing and gillnetting. In total, 923 fish from seven species were sampled, primarily juvenile Atlantic salmon and brown trout. One salmonid was found to be IPN positive by tissue culture.

Nodaviridae:

Nodaviridae are considered the most economically important emerging viral pathogen and halibut, cod, and salmon are considered susceptible (Bruno and Woo, 2002). The Nodaviridae have been reported as the causative agents of disease throughout the world and referred to as viral nervous necrosis (VNN) or viral encephalopathy and retinopathy (VER). Lesions show the classic signs of VNN and the brain and retinal tissues from moribund fish showed diffuse degenerative vacuolative encephalopathy and degenerative histiocytic retinitis. Significant losses are reported in juvenile and adult fish. In Norway, mortality reaching 100% has occurred in juvenile farmed Atlantic halibut (Grotmol *et al.*, 1997; Johans *et al.*, 2002), and Aspehaug *et al.* (1999) reported viral nervous necrosis in adult and mature halibut. In addition, mortality between 11–60% of farmed sea bass (*Dicentrarchus labrax*) at different locations in Greece has been reported (Le Breton *et al.*, 1997). Disease outbreaks have also occurred in juvenile cod (Starkey *et al.*, 2001), hence hatcheries must reduce the risk of this virus entering if reliable larval fish production is to be maintained; however, vertical transmission of the virus may hinder control measures. Recently RT-PCR in broodstock cod were found in the USA, although no virus was isolated, and large-scale mortality of cultured haddock occurred in a facility in New Brunswick, Canada. VER continues to be a problem in sea bass farms in France and a new isolate was found in sea bass. It was pathogenic at temperatures < 18 °C and was similar to the isolate of Atlantic halibut. There were no isolations of nodavirus from wild or farmed fish in Scotland during 2002, but the increasing number of susceptible species identified worldwide will increase the chance of infection moving from wild to farmed stock and vice versa.

Infectious salmon anaemia virus (ISAV):

Infectious salmon anaemia virus (ISAV) is the cause of ISA in salmonids, and is a serious problem in farmed fish in Norway, Scotland, Canada, Maine, USA, and the Faroe Islands. The virus has been shown experimentally to be transmitted between fish in fresh water, and experimentally and clinically in sea water. The infection is spread by management activity such as well-boat traffic, but also possibly through contact with wild fish (Rimstad and Mjaaland, 2002).

The susceptibility of cod and halibut to ISAV infection is unknown, although experimentally it has not been possible to infect Atlantic halibut or Atlantic cod, indicating that they are resistant to this virus. In addition, wild-caught saithe (*Pollachius virens*) appear to be resistant to the Norwegian isolate of ISAV and incapable of supporting its replication (Snow *et al.*, 2002). Hence, saithe that co-exist with salmon in and around aquaculture facilities are unlikely to have a significant impact on the epizootiology of this virus. In addition, the apparently limited host range for ISAV would suggest that outbreaks will be infrequent in new fish groups. A wild host for the virus has not been identified.

Infectious haematopoietic necrosis virus (IHNV):

Atlantic salmon are the primary species cultured in British Columbia, Canada, and are particularly susceptible to IHNV. Several farms experienced significant losses in the winter of 2001/2002. The source of the virus is suspected to be wild stocks in the vicinity of the Atlantic salmon farms since all fish are health-tested prior to transfer to the net cage sites.

Viral haemorrhagic septicaemia virus (VHSV):

Viral haemorrhagic septicaemia (VHS) is traditionally considered a disease of rainbow trout resulting in extensive losses to freshwater culture operations across continental Europe. An outbreak of VHS in farmed turbot in Scotland in 1994 prompted the initiation of a widespread VHS virus (VHSV) surveillance programme across the European marine environment. This resulted in the recovery of over 150 isolates of VHSV from a wide range of wild marine fish species in the North Sea, Baltic Sea, and eastern Atlantic (Mortensen *et al.*, 1999; King *et al.*, 2001). Most isolates are from herring (*Clupea harengus*) and sprat (*Sprattus sprattus*). It is likely that the Baltic Sea, the Kattegat, and the Skagerrak are the areas with the highest prevalence of marine VHS in Northern Europe. Studies show that marine VHSV isolates are of low risk to rainbow trout and Atlantic salmon and marine VHSV isolates appear to be of low risk to marine aquaculture species other than turbot. VHSV has been reported to infect wild cod, haddock, and many other marine fish species (Mortensen *et al.*, 1999; Smail, 2000; King *et al.*, 2001), although epizootics resulting in morbidity have not been observed in wild fish in Europe.

In 1988 adult chinook and coho (*Oncorhynchus kisutch*) salmon returning to coastal hatcheries in Washington State, USA, were examined and found to harbour VHSV. This resulted in the mandated destruction of all fish (approximately 5 million eggs, fry, and juveniles) and complete disinfection of the two facilities (Winton *et al.*, 1991). In subsequent years, VHSV was recovered from other stocks of salmon in Washington State and from Pacific cod in Prince William Sound, Alaska (Stewart *et al.*, 1990; Meyers *et al.*, 1992). Research comparing the North American and European isolates of VHSV revealed differences at the biological level, such as growth in cell culture (Batts *et al.*, 1991) and virulence (Winton *et al.*, 1991). Additional differences were found at the genetic level (Oshima *et al.*, 1993). A European strain of the viral haemorrhagic septicaemia virus (VHSV) isolated from wild-caught cod was shown to cause clinical disease and mortality in excess of 80% in juvenile Atlantic cod when administered by the intra-peritoneal (i.p.) route (Snow *et al.*, 2000). Cod were not susceptible to VHSV following waterborne exposure and this raises important questions surrounding the propagation, maintenance, and impact of a naturally occurring reservoir of virus in the marine environment. Furthermore, Atlantic halibut appear to be resistant. Recently, high mortality has been reported in sardines (*Sardinops sagax*) around Vancouver Island, Canada, with isolation of the European strain. The sandeel (*Ammodytes* sp.) was also recently identified as a carrier of VHS in Denmark. VHSV is now divided into several genotypes and the risk of infection and transfer will depend on the species and genotype.

Aquareovirus:

An Aquareovirus was demonstrated in macrophages of both naturally and experimentally infected halibut in Canada (Cusack *et al.*, 2001). The virus survived and replicated in the phagocytic cells, as demonstrated by electron microscopy. In Scotland, farmed moribund Atlantic halibut larvae were sampled for a suspect nodavirus infection. An examination of the tissue sections showed moderate degeneration of hepatocytes with slight haemorrhage or capillary dilation, irregular clear vacuoles, cell rounding, and early necrosis with cell fragments. Kidney tubule dilation was recorded and an increased faint eosinophilic staining within the trabeculae layer. The samples were negative for IPN, VHS, ISA, nodavirus, and bacterial infection and currently the aetiology of the mortality in Scotland is a suspect Aquareovirus.

Sleeping disease virus (SDV):

An outbreak of sleeping disease was confirmed at a trout fish farm on the River Earn, Scotland system in June 2002. Wild fish were obtained from three locations within the catchment area, using a combination of electrofishing and gillnetting. In total, 176 fish consisting of juvenile Atlantic salmon, juvenile and adult brown trout, escapee rainbow trout, stone loach (*Barbatula barbatula*), and minnow (*Phoxinus phoxinus*) were sampled. Tissue culture, PCR, histology, and blood samples were obtained and the tests were negative. No isolations have been found in the wild fish samples.

Salmon pancreas disease virus (SPDV):

Salmon pancreas disease prevalence is low in farmed salmon and the virus has not been detected in wild fish in Scotland. Weston *et al.* (2002) reported minor biological differences between SPDV and SDV and concluded on the basis of the close genetic similarity that they are closely related isolates of the same virus species for which the name Salmonid Alphavirus was proposed. However, there is increasing mortality at many sites in Ireland. The disease has been observed at low levels during the past 8–9 years and its re-emergence in 2002 cannot be readily explained. Certain sites have suffered in excess of 30% stock losses, whilst others have experienced lower but more protracted losses. A nationwide epidemiological study has just been launched to determine the factors which may have contributed to the re-emergence of this economically significant disease. Pancreas disease was diagnosed at twelve sea sites in Norway. Two of these cases were in rainbow trout. The losses vary, but may last over a long time period and become substantial. All reported cases so far have been in western areas, however, there is a trend that the disease is spreading. Disease cycles and incidence will change over time and at the present time this disease is restricted to farmed fish.

Bacterial infections

Resistance and susceptibility vary among the major bacterial pathogens affecting farmed halibut, salmon, cod, and trout. For example, halibut are resistant to *Aeromonas salmonicida*, hereas *Renibacterium salmoninarum* is confined to trout and salmon. Bergh *et al.* (2001) concluded that *Vibrio anguillarum*, *Flexibacter ovolyticus,* and atypical *Aeromonas salmonicida* were the major bacterial pathogens affecting halibut production.

Wild fish have been recorded dying of furunculosis during high water temperatures following transfer. In Norway, furunculosis has been a problem for farmed fish since its first occurrence in 1964, following an introduction of rainbow trout from Denmark. The disease spread to several farms and also to wild stocks thereafter. However, eradication programmes were successful. In 1985, the disease was again recorded in marine salmon farms, following importation of salmon smolts from Scotland. By the end of 1992, furunculosis had spread to 550 farms and concurrently to 74 natural watercourses, where it affected wild salmon, sea trout, and brown trout. Although the ecological consequences are not known, there is concern that the disease may affect populations if a significant number of the broodstock die before spawning.

Between 1998 and 2002 there has been no major trend in bacterial infections. It is noteworthy that *Piscirickettsia salmonis* was confirmed in Scottish Atlantic salmon from tissue sections and enzyme-linked immunosorbent assays (ELISA) in 2002.

Parasite infections

Protozoan and metazoan parasites have been recorded from halibut and include *Ichthyobodo* sp., the microsporidium *Enterocytozoon* sp., *Trichodina hippoglossi*, *Entobdella hippoglossi*, and the parasitic copepod *Lepeophtheirus hippoglossi* (Bergh *et al*., 2001). In general, the impact of these parasites on farmed and wild fish is unknown.

Sea lice (*Lepeophtheirus salmonis*):

Norway started a National Action Plan against "Salmon lice on Salmonids" in 1997. The legal limit for the maximum mean number of lice per farmed fish, strategic regional treatments against lice, and compulsory reporting of lice numbers were included (Heuch *et al*., in press). Bag net catches show that the homing salmon are infected by juvenile lice as the fish pass through coastal zones and this appears to be related to salmon farming. Homing salmon that stay in fjords and coastal areas for prolonged periods may have a large number of mature female lice. Work by Karasev *et al.* (1997) shows that salmon lice infestations in the White Sea basin show low salmon lice in this farm-free area. Data show high salmon lice infestation levels in populations of sea trout and sea charr (*Salvelinus* sp.), and predictions of negative effects on fish populations. Long-term experiments and monitoring are required to assess the regulatory effect of lice in these populations. Sea trout and Arctic charr (*Salvelinus alpinus*) over-winter in fresh or brackish waters and the lice generally die, resulting in a low prevalence and intensity of adult females; the infection pressure is low and does not constitute a threat to running salmon smolts. Heuch *et al.* (2003) has looked at national data sets from Scotland and Norway and concluded that the salmon in Scotland are exposed to a larger number of parasites than Norwegian farmed salmon, in particular regarding the chalimus stage. As possible causes, the authors discuss the use of shallower and more enclosed water bodies for farming, smaller and shallower pens, or differences in sea water temperature or in access to appropriate medication in Scotland.

***Gyrodactylus salaris*:**

Gyrodactylus salaris has occurred in three salmon hatcheries in Norway. One of the hatcheries was infected in the 1970s and remained clear until now. The hatchery is located close to an infected river system and could represent the source of this infection. Infections in the other hatcheries could have resulted from fish movements.

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8.5 Effectiveness of Salmon Farming Management Control Methods for Sea Lice in ICES Member Countries

Request

This is part of continuing ICES work to consider new developments with regard to the control of fish and shellfish diseases.

Source of the information presented

The 2003 report of the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) and ACME deliberations.

Summary

Infestation with sea lice (*Lepeophtheirus salmonis*) is a major problem for the salmon farming industry, with a high risk of transfer to wild stocks (salmon and sea trout). There is therefore a great need for ICES to provide advice on the most recent and effective control methods for sea lice.

This section reviews practices applied in Scotland and Norway for the prevention and control of sea lice in salmon farms. From the information available, it can be concluded that the most successful approach to reduce the number of lice seems to be "integrated pest control" involving a farm management regime that includes both preventative and treatment strategies.

Recommendations and advice

Owing to the continuous significance of sea lice for salmon farming and the risk of transfer between farmed and wild fish, ICES recommends that relevant Member Countries intensify studies on the behaviour and dispersal of infective larval stages of sea lice, in order to better understand the dynamics of the parasite infestation.

Furthermore, ICES recommends that relevant Member Countries continue efforts to develop new, and to improve existing, preventive measures. These include coordinated management activities (use of chemotherapeutants, coordinated fallowing, and relocation of cages), an improvement of the host immune response, and effective vaccines.

Scientific background

Sea lice infections have a great impact on salmon farms and infections are transmitted from wild to farmed fish and vice versa. Grazing of the parasite on the mucus stresses the fish, and leads to epidermal erosion and eventually death of the fish. Furthermore, sea lice may transmit other pathogens and may infect wild fish.

Salmon lice cost the Scottish salmon farming industry about £30 million per year in treatment costs and production losses. Current chemical control methods are sometimes inadequate and are contentious in terms of environmental and human health. However, despite intensive research on sea lice biology and control on salmon farms, the factors influencing re-infestation rates from other farms and wild fish remain poorly understood, and sea lice remain the most economically significant parasite in salmon farming in sea cages. In this context, models have been developed predicting the dynamics of the parasite population and the dispersal of infective stages of the parasite. However, work on these models has to be continued in order to consider, to a greater extent, the factors affecting population growth and parasite dispersal, such as hydrodynamics.

A growing number of therapeutants are being used to control lice abundance but this does not necessarily prevent their occurrence. Fallowing/site rotation is considered an effective method of prevention on cage sites, although there is evidence that new infections from wild fish or escapees can occur. Furthermore, tidal transport of the larval stages within and between sites may lead to re-infestation.

In Norway, a "National Action Plan" against sea lice was developed in 1997. This plan contains the following objectives:

1) to establish regional working groups to plan and coordinate regional efforts;
2) to document strategic lice treatments;
3) compulsory reporting of lice numbers;
4) compulsory reporting of lice treatments; and
5) monitoring salmon lice infections on sea trout and wild salmon smolts.

The approach in Scotland to the sea lice issue has been presented in the recently published document "A Strategic Framework for Scottish Aquaculture", which is available at http://www.scotland.gov.uk/library5/environment/sfsa-00.asp. The integrated management strategy, which has already been adopted within Scotland, includes:

a) synchronous year-class stocking within defined management areas;
b) regular and accurate sea louse counts;
c) effective treatments to achieve zero, or near zero, ovigerous sea louse levels.

In addition, Scotland will, where practicable, adopt novel techniques to reduce reliance on chemotherapeutants.

Differences exist in lice numbers per fish between Norway and Scotland. Possible causes, including the use of shallower and more enclosed water bodies for farming, smaller and shallower pens, or differences in sea water temperatures or in access to appropriate medication in Scotland, have been discussed (see Section 8.4, above).

Biological control methods, such as improving the host immune response and developing a vaccine, may be preventative if they are successful. The use of wrasse as cleaner-fish can be effective and has been widely employed in Norway. These biological approaches may significantly reduce the need for medicines. However, it can be concluded that no one method of prevention or control is likely to be universally successful. The most successful approach to reduce the number of lice seems to be "integrated pest control" involving a farm management regime that includes both preventative and treatment strategies.

9 INTRODUCTIONS AND TRANSFERS OF MARINE ORGANISMS

9.1 Code of Practice on Introductions and Transfers of Marine Organisms 2003

Request

This is part of continuing ICES work to review and update the 1994 Code of Practice on the Introductions and Transfers of Marine Organisms and provide technical guidelines on implementation.

Source of the information presented

The 2002 and 2003 reports of the Working Group on Introductions and Transfers of Marine Organisms (WGITMO) and ACME deliberations.

Summary

Work has continued in 2002/2003 to provide an updated Code of Practice on Introductions and Transfers of Marine Organisms, with particular emphasis on recent concerns regarding introductions of genetically modified organisms, and to increase awareness of the risks associated with trade in aquarium species, as well as other species which are part of current commercial practice. The revised Code of Practice is attached as Annex 7.

Technical Guidance notes on the contents of a prospectus for a proposed introduction, risk assessment, quarantine, and monitoring and a flow chart have been developed and will be provided on the ICES website. A case study on the history of the introduction of *Patinopectin yessoensis* in Ireland, which serves as a model of a previous introduction, will also be provided on the ICES website.

Recommendations and advice

ICES recommends that Member Countries adopt and apply the revised Code of Practice on Introductions and Transfers of Marine Organisms, 2003, for reaching decisions on new introductions of species, for introducing or transferring species which are part of commercial practice, and for reaching decisions regarding new releases of Genetically Modified Organisms (GMOs).

ICES also draws the attention of the OSPAR Commission and the Helsinki Commission to this Code of Practice, for potential use in their regulatory activities.

9.2 Current Status of Deliberate Fish, Shellfish, Algal, and Other Introductions in and between ICES Member Countries

Request

ICES Member Countries may request ICES to review proposed deliberate 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 2003, but the status of ongoing introductions and transfers was reviewed.

Source of the information presented

The 2003 report of the Working Group on Introductions and Transfers of Marine Organisms (WGITMO) and ACME deliberations.

Recommendations and advice

ICES notes that this information is of interest to Member Countries and encourages them to submit National Reports (including the outcome of planned introductions) on an annual basis.

Scientific background

The ACME reviewed the WGITMO report and agreed to present the information contained below.

Trade in live aquatic organisms within and between ICES Member Countries continues mainly for aquaculture, restocking, live food sales, and ornamental pet fishes. This trade is summarized in Table 9.2.1. The most commonly moved species in 2002 continue to be Atlantic salmon (*Salmo salar*) and Pacific oysters (*Crassostrea gigas*).

The information in the table is not fully comprehensive as not all ICES Member Countries submitted National Reports to WGITMO. Further, the origin of several importations remains unclear as some countries exhibit a lack of import and/or export documentation.

Table 9.2.1. Summary of live imports and exports of aquatic species according to National Reports submitted to WGITMO 2003. Ornamental trade excluded. (Abbreviations: cr = crustacean, fi = fish, mo = molluscs, pl = plants, Aus = Australia, Bel = Belgium, Can = Canada, Cze. R = Czech Republic, Den = Denmark, Est = Estonia, Fin = Finland, Fra = France, Ger = Germany, Hun = Hungary, Ice = Iceland, Ind = Indonesia, Ire = Ireland, Ita = Italy, Lat = Latvia, Net = the Netherlands, Nor = Norway, Pol = Poland, Por = Portugal, Rus = Russia, S. Afr = South Africa, Spa = Spain, Swe = Sweden, UK = United Kingdom, USA = the United States of America.)

	Import (limited to ICES Member Countries)													
Exporting country	**Bel**	**Can**	**Est**	**Fin**	**Fra**	**Ger**	**Ire**	**Net**	**Nor**	**Pol**	**Rus**	**Swe**	**UK**	**USA**
Australia							fi							
Belgium														
Canada					fi	cr fi		fi		fi		cr mo	cr	fi
Czech Republic						fi								
Denmark			fi			fi mo	fi		fi			fi mo	fi	
Estonia								fi	fi		fi		fi	fi
Finland			fi								fi			
France						fi	fi mo					mo	fi	
Germany	mo							mo				fi		
Hungary						fi								
Iceland		mo					fi		fi					
Indonesia							pl							
Ireland					fi mo	cr fi						cr	fi mo	
Italy					mo	fi						fi		
Netherlands					mo	fi						fi mo	fi	
Norway						cr						cr fi mo		
Poland						fi								
Portugal														
Russia						fi								
Spain					mo				fi					
S. Africa													fi	
Sweden						fi								
UK			fi		cr mo		fi mo		fi			fi mo		
USA		fi mo										cr mo		

Two examples of proposals for deliberate introductions that are currently under review are provided below.

Proposal for an inter-basin transfer of sturgeons

The Society to Save the Sturgeon (SSS), a non-profit organization founded in 1994 aiming at the restoration of the sturgeon populations in Central Europe, currently is carrying out a project for the restoration of sturgeon populations in German waters. This proposal was presented at the 2003 meeting of WGITMO.

Recent evidence from archaeological, morphological, and genetic studies has proven that the North American Atlantic sturgeon (*Acipenser oxyrinchus*) became established in the Baltic Sea and its tributaries approximately 1000 years before present (b.p.). These surprising results were discussed at an international workshop in June 2002. The recommendation of the workshop provided the basis for a decision by the SSS and the German Federal Agency for Nature Conservation, which is hosting the project, to diversify the initial attempt to protect and re-establish *Acipenser sturio* in Germany. The current plan is to commence the restoration plans for the Baltic with *A. oxyrinchus* and for the North Sea and its tributaries with *A. sturio*. Therefore, the SSS proposed a plan to transfer broodstock of North American *A. oxyrhinchus* under the new Code of Practice as a basis for the future restoration activities for the species in the Baltic Sea.

Proposed introductions of the Asian *Crassostrea ariakensis* into tidal waters of the Chesapeake Bay

A proposed introduction of the Asian oyster *Crassostrea ariakensis* into tidal waters of the Chesapeake Bay is being advanced at an experimental scale to further test the performance of these oysters and to evaluate commercial feasibility. The Virginia Seafood Council has developed a plan to grow approximately 1 million oysters among ten specific commercial shellfish operations in the lower Chesapeake Bay. This plan, which calls for the use of triploid oysters, has undergone two panel reviews by the Chesapeake Bay Program and is the focus of a study by the National Research Council. The planned introduction is highly controversial, due to the desire to revive a once-productive oyster fishery in the Chesapeake Bay and the potential risks associated with the use of a non-native species. The decision and legal authority to proceed reside with the State of Virginia, and it appears that the field "trials" of triploid oysters will commence in summer 2003; any oysters used in such trials are to be monitored over time and removed prior to reaching maturity.

The Chesapeake Bay has historically supported a large commercial oyster fishery, based upon the native oyster *Crassostrea virginica*. Since the late 19th century, production of the native oysters has declined dramatically. Today, the oyster fishery harvest is approximately 1% of its historical peak. The decline in oyster harvest has had a large economic and social impact in the region, and is also thought to have resulted in major shifts in the foodweb, chemical cycling, and nursery habitats for many organisms (see Kennedy *et al*., 1998, and references therein for discussion).

The decline in the native oyster population (standing stock) is attributed to a combination of factors, including overfishing, habitat alteration, sedimentation, and diseases (Kennedy *et al*., 1998). A large effort exists within the region to restore the native oyster population and a viable commercial fishery (Leffler, 2002). It appears that diseases are the primary deterrent to recovery. The native oyster is infected by two protistan parasites, *Haplospordium nelsoni* (MSX) and *Perkinsus* spp. (Dermo), which can cause very high (>90%) mortality in some years. Both parasites emerged as a major source of mortality in the mid-20th century. The reason for the emergence of both diseases was unclear, and both parasites were new to science when first discovered. More recently, genetic analysis indicates that one of the parasites (MSX) is not native to the region, and was introduced with the Asian oyster *Crassostrea gigas* (Burreson *et al*., 2000), although this non-native host oyster did not become established.

Simultaneous with current efforts to restore the native oyster, there has been some exploration about the possible use of a non-native oyster to restore an oyster fishery to the region. In the past few years, this effort has focused on another Asian oyster, *Crassostrea ariakensis*. Various lines of research suggest that this oyster could perform well in waters of the Chesapeake Bay, exhibiting relatively high rates of growth and survivorship under local environmental conditions, even when challenged with the local parasites (Calvo *et al*., 2000).

Based upon these results, a plan to test the commercial feasibility of this oyster in Virginian waters is being advanced. The plan calls for the introduction of approximately 1 million juvenile oysters for grow-out at multiple sites in the lower Chesapeake Bay. This plan calls for the use of triploids in the field-based trials to minimize the chances of reproduction and establishment of feral populations.

Various concerns exist about the uncertainty associated with this pilot introduction, as well as the potential for a rapidly expanding population (Thompson, 2001; Chesapeake Bay Programme, 2002, 2003; Leffler, 2002). These concerns are highlighted in a recent review by Leffler (2002), as follows:

- First, the initial plan called for the use of triploids created by chemical treatment, whereby a subset of the oysters would remain diploid and others could revert to this state. A revised plan calls for the use of mated triploids. Reproduction and population establishment of the oyster, using this approach, is a possibility. Should this occur, no one can predict whether this non-native oyster would affect the native oyster (or other species) through a variety of mechanisms, whether it would have a similar or different functional role compared to the native oyster, and how it may alter the Chesapeake Bay ecosystem. This species appears physiologically capable of spreading outside of the Chesapeake Bay, from New England to the Caribbean. However, little is known about the biology and ecology of *C. ariakensis* in its native range or elsewhere, to provide clues (let alone robust predictions) about the possible consequences of introduction.
- Second, independent of the oyster itself, the possibility of introducing other organisms exists. The proponents have followed recommended protocols for intentional introductions of marine organisms under the 1994 Code of Practice on the Introductions and Transfers of Marine Organisms, including the use of at least second-generation organisms that were reared in the laboratory and screened for known pathogens. This approach serves to prevent the transfer of many organisms, which are associated with the initial imports. However, the screening is limited to known pathogens, and little is known about this oyster. The identity and effects of most microorganisms, some of which are transferred vertically from parent to offspring, remain unknown.

Thus, despite these protocols, some microorganisms from the original source are likely to be introduced with *C. ariakensis*. It remains a challenge to assess the potential consequences of such transfers of microorganisms into the Chesapeake Bay.

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9.3 Selected examples of Current Accidental Invasions, their Consequences and Significance

Request

This is part of continuing ICES work to keep under review new information concerning accidental introductions and movement of non-indigenous marine organisms.

Source of the information presented

The 2003 report of the Working Group on Introductions and Transfers of Marine Organisms (WGITMO) and ACME deliberations.

Recommendations and advice

Noting with concern the serious ecological and economic implications of the accidental introductions of non-indigenous species described here, ICES urges all Member Countries to submit national reports on an annual basis describing accidental species invasions and, where possible, the consequences of these invasions.

Scientific background

The ACME noted that accidental introductions and transfers of non-native species continue to be documented in many ICES Member Countries. Three examples are described below.

Club tunicate (*Stylea clava*) on the Atlantic coast of Canada

The club tunicate is indigenous to Asia and was introduced to the Pacific coast of North America in the early 1920s. It was first reported in Georgetown Harbour, Prince Edward Island, in 1999. The tunicate has since spread to the Montague River, Brudenell River, St. Mary's Bay, Murray River, Malpeque Bay, and the Orwell/Vernon area—all major mussel production areas of PEI. The initial introduction is believed to have occurred with commercial ship movements in Georgetown Harbour. Subsequent spread may have been through natural dispersion, fouling of fisheries gear, leisure boating, and aquaculture activities. To reduce the further spread of this exotic tunicate, transfers of all shellfish from affected areas have been restricted through the Introductions and Transfers Committee process. The tunicate is causing serious economic losses to the mussel production industry of PEI and, if uncontrolled, has the potential to spread and significantly impact biodiversity and mussel production throughout the southern Gulf of St. Lawrence. Since the time it was introduced, it has spread to several economically significant mussel production areas of PEI, including St. Mary's Bay, Murray River, and Malpeque Bay. The club tunicate reduces mussel productivity and increases the costs associated with defouling on site, as well as at the processing plant. The tunicate also impedes seed collection and relay for grow out, as well as transfers for open-effluent processing. Another impact is the seasonal die-off of the tunicates and associated environmental degradation.

Slipper limpet (*Crepidula fornicata*) in France

The slipper limpet *(Crepidula fornicate)* has been introduced by stages. The most important event was in the 1970s, associated with the introduction of the Japanese oyster (*Crassostrea gigas*) following high mortality of Portuguese oysters (*Crassostrea angulata*). A research programme (LITEAU), funded by the French Ministry of the Environment, was developed in 2000 for a three-year period to better understand the species proliferation mechanisms and the impacts on the marine environment and, particularly, the commercial shellfish industry. At the end of the project, the main results are:

1) The biomass of *Crepidula fornicata* was evaluated in four pilot sites: the Bay of Arcachon (155 metric tonnes), the Bay of Marennes Oléron (5,000 metric tonnes), the Bay of Brest (18,500 metric tonnes), and the Bay of St Brieuc, where the colonization is recent but the stock is the most important with 250,000 metric tonnes. These differences should be related to dredging and trawling activities, which modify the sea bottom by making lines where *C. fornicata* populations could find good conditions to increase;
2) With an efficient reproductive strategy and a ten-year longevity, *C. fornicata* is a sustainable invasive species;
3) A spatial competition exists between this species and commercial shellfish (e.g., scallops), but the feeding competition is low or not direct.

Ectoparasite *Gyrodactylus salaris* in Sweden

The introduced species receiving the most public attention during 2002 in Sweden was the ectoparasite *Gyrodactylus salaris* (trematode skin fluke) found in 2001 on rainbow trout farmed in cages in the Lake Södra Bullaren (province of Västergötland close to the Norwegian border). The lake discharges into two Swedish/Norwegian rivers with salmon (Enningdalsälven and Långevallsälven). *Gyrodactylus* can survive for some time in brackish waters, but only for a very short time in marine waters. The parasite is native in the eastern part of Sweden, where the salmon populations are little affected. However, it has caused declines in some salmon populations on the Swedish west coast, where it turned up during the late 1980s. It is considered a great problem in Norway, where native salmon die or often are heavily affected (more than 10,000 parasites have been found on one single young fish), although some salmon may survive and get rid of the infection, as do salmon from the Swedish Lake Vänern. During 2002, the Fishery Board of Sweden came out with recommendations to sport fishery associations to clean equipment and boats before being used in new waters and not to clean out fish in other waters nor to move them or water to other sites. This case went all the way up to the ministries and Norwegian activists also came in and slaughtered farmed fish. It was later recognized through laboratory studies that the Norwegian salmon populations in those rivers were sensitive to attacks from the parasite and the Swedish county council finally closed the farm site and forbade further farming of salmonoid fish at that site after 1 December 2002.

9.4 Progress in Ballast Water Research and Management Technologies

Request

This is part of the continuing ICES work to keep under review new information concerning research and management of ballast water and other ship vectors for transfers of marine organisms.

Source of the information presented

The 2003 report of the ICES/IOC/IMO Study Group on Ballast and Other Ship Vectors (SGBOSV) and ACME deliberations.

Recommendations and advice

ICES notes the international concerns surrounding the development and implementation of effective and environmentally sound ballast water management and treatment methods. There is a need for prioritized funding to develop and implement international standards and international procedures for the evaluation and approval of new ballast water treatment systems.

Summary

The ongoing concern and the evidence for the global spread of invasive species by ballast water, sediments, and ships' hulls have been the subject of discussion by SGBOSV for a number of years. At its 2003 meeting, SGBOSV reviewed a number of papers of relevance to these vectors for the spread of invasive species. Of particular note is the Ballast Water Treatment R&D Directory (http://globallast.imo.org/research) produced and maintained by the IMO GloBallast Programme, which provides a comprehensive directory of projects relevant to the introduction of invasive marine species via ships and potential management measures.

Several papers reported direct results from current ballast water treatment R&D projects, including 1) the effectiveness of ballast water exchange at sea, 2) cyclonic-UV treatment, 3) chemical treatment with PeraClean ocean, 4) three new projects currently under development in Germany that will review various technologies, 5) the EU MARTOB (On-Board Treatment of Ballast Water and Application of Low-Sulphur Marine Fuel) project which is also reviewing various treatment technologies, and 6) a potential new concept based on a steam-driven high velocity pumping system.

Two papers reported on the development of test protocols and procedures for the evaluation, acceptance, and approval of alternative ballast water management and control measures and treatment technologies, specifically the U.S. EPA Environmental Technology Verification (ETV) Program and the German *Artemia* Testing Protocol. It was made clear that the current lack of internationally accepted testing and evaluation procedures is a major gap, and that procedures to ensure the objectivity, scientific validity, statistical rigour, and quality control of data from treatment technology testing is an urgent need.

A paper reported on the status of the IMO Ballast Water Convention and the draft ballast water treatment standards contained in the draft Convention. It was made clear that finalization of these standards is vital so as to provide the R&D community with a clear benchmark to aim for in developing alternative treatment technologies.

Conclusions

The ACME agreed to the following general conclusions in relation to this issue:

1) Currently implemented ballast water control and management measures are restricted to ballast water exchange at sea. All other treatment methods must still be considered experimental and developmental only, and significant further R&D is required before any alternative treatment technique can be accepted and approved by regulatory agencies.
2) Ongoing research into ballast water exchange at sea continues to indicate that this method is highly variable in biological effectiveness. In some cases, it results in increases in the abundance of organisms and/or the diversity of species in ballast tanks. This method continues to suffer from significant limitations, including restricted usefulness for ships on coastal and/or shorter voyages. Further, safety concerns have been expressed regarding undertaking ballast water exchange at sea.
3) There is an urgent need to find alternative, more effective, and environmentally sound ballast water treatment methods.
4) It appears that any new ballast water treatment system will involve a combination of technologies, for example, primary filtration or physical separation followed by a secondary biocidal treatment.
5) There is a need to develop and implement international standards and international procedures for the evaluation and approval of new ballast water treatment systems. The U.S. EPA ETV process and the German *Artemia* Testing Protocols provide very good models for consideration for adoption or adaptation as the basis for such international procedures.
6) The GloBallast R&D Directory and R&D Symposium series are very useful resources and frameworks for the ongoing review, updating, and communication of progress on these issues at the global level.

9.5 Directory of Dispersal Vectors of Exotic Species

Request

This is part of ongoing ICES work to keep under review and report on pathways for the introduction and transfer of exotic species in the marine environment.

Source of the information presented

The 2003 report of the Working Group on Introductions and Transfers of Marine Organisms (WGITMO) and ACME deliberations.

Scientific background

During the past few years, WGITMO has worked on the preparation of a Directory of Dispersal Vectors (entitled "Vector pathways and the spread of exotic species in the sea"). This document outlines the principal vectors that are likely to result in the further spread of non-indigenous species, including both introductions and translocations. The document reviews the current state of knowledge concerning various vectors of species introductions, provides a brief overview of the potential risks associated with each category of vector, and identifies significant knowledge gaps. Although there is a reasonable understanding of the vectors, assigning vector strengths can be difficult and largely depends on local or regional trading activities, and political and socio-economic circumstances. Not all vectors continue to operate and some become more significant at specific times.

Some vectors may transport fundamentally different sets of organisms (e.g., mussels attached to a ship's hull, species nestling within hull fouling or on fouling organisms, species burrowing into mussel shells, and pathogens or microalgae inside the mussels). Conversely, some species may be spread by several different vectors (e.g., larval mussels may be transported in the ballast water; adult mussels as hull foulers, in intentional movements for aquaculture, or as accidental aquaculture associates).

The Directory of Dispersal Vectors, "Vector pathways and the spread of exotic species in the sea", will be published in the *ICES Cooperative Research Report* series.

10 MARINE BIOLOGICAL COMMUNITIES, PROCESSES, AND RESPONSES

10.1 Benthos Issues

10.1.1 Progress in the North Sea Benthos Project

Request

This is part of continuing ICES work to support activities and implement projects on the monitoring of the marine environment and biological communities within the ICES area.

Source of the information presented

The 2003 report of the Benthos Ecology Working Group (BEWG) and ACME deliberations.

Summary

Among other activities, the BEWG reported on the progress in the North Sea Benthos Project (NSBP), with special attention paid to quality control, data analyses, workshops, and end products. This project involves a combination of re-sampling of stations from the 1986 ICES North Sea Benthos Survey (NSBS) and the compilation of data from other relevant sources collected in the period 1999–2001. Three workshops have been held since the initiation of the project, with the principal aim of addressing problems associated with the inter-compatibility of data from different sources. The hypotheses assume the occurrence of changes in the status of benthic assemblages in parts of the North Sea between 1986 and 2000, and that the magnitude of these changes are sufficient to warrant further explanation. Several questions will be addressed by the NSBP in the future, on the causative factors of changes in the benthic assemblages.

Recommendations and advice

ICES recommends that, in the continuation of the North Sea Benthos Project, a key hypothesis be tested on the role played by environmental changes in the fluctuations of biological communities. Similar initiatives should be encouraged for studies of other biological communities (phytobenthos, phytoplankton, and zooplankton).

ICES recommends that Member Countries devote increased attention to the ecology and biology of key species, in order to better understand and assess their sensitivity to environmental changes in terms of life cycle, reproduction success, and spatial distribution.

Scientific background

From different sources, it appears that changes in the characteristics of biological communities are difficult to correlate with major environmental fluctuations in relation to anthropogenic influences (eutrophication and contaminants) and large climatic trends, as they are often described by the North Atlantic Oscillation (NAO) index. The North Sea Benthos Project constitutes an opportunity to test hypotheses related to the action of environmental changes on biological communities.

The main objectives of the North Sea Benthos Project are identified as follows: Are changes an artefact of sampling and/or analytical practices between laboratories or countries? Can changes be ascribed to the effects of eutrophication (including amelioration in conditions)? Can changes be ascribed to alterations in the pattern and intensity of demersal fishing between 1986 and 2000? Can changes be linked to climatic trends or cycles such as increased water temperatures or storminess (including variation in the NAO index – see explanation in Section 6.6.1)? Can changes (especially in coastal environments) be linked with changes in the nature and quantities of contaminant/organic matter inputs?

Special attention will be devoted to the changes occurring within similar habitat types, and to measures of assemblage structures, such as the occurrence of rare species and their sensitivity to natural or man-made influences. The interpretation of differences between the results of NSBP 2000 and NSBS 1986 results should benefit from the use of long-term data sets on benthos: monitoring data for benthos from 1970 to 2000 should be available from Belgium for the NSBP.

Initial multivariate analyses of the incomplete data set were conducted at the March 2003 NSBP workshop and, while in-depth interpretations of the output at this stage would be inappropriate, there are encouraging indications that interlaboratory inconsistencies did not dominate patterns.

10.1.2 Results of statistical analyses of benthos community data

Request

This is part of continuing ICES work to support research and collect information on benthic communities and their changes, and to monitor long-term trends using biological measurements.

Source of the information presented

The 2003 report of the Working Group on Statistical Aspects of Environmental Monitoring (WGSAEM) and ACME deliberations.

Summary

The results that were analysed are based on time series data on benthic macrofauna from the HELCOM database, and the National Environmental Research

Institute, Denmark. Data were analysed using Multidimensional Scaling (MDS) and other techniques (taxonomic distinctness).

Examples of correlations are given for three stations, from which it can be inferred that the match is rather modest in most cases. The influence of other environmental variables, not included in this study, and/or spatial heterogeneity of the macrofauna may be inferred in relation to these low matches.

The average taxonomic distinctness of the macrofauna, measuring the average breadth of a sample, is a very intuitive definition of biodiversity. It remains independent of the number of species within the sample. It has been computed on the same biological data. The taxonomic distinctness during the time of observation could not be related to particular environmental variables. However, because the taxonomic distinctness is independent of the size of the sample, it could be useful in the analysis of historical data when sample sizes were not comparable to those taken at present.

Recommendations

ICES recommends that ordination techniques be used with care in the analysis of benthic community data; specifically, they can be used for detailed analyses of specific data sets, when local expertise can be used to improve both the data analysis and the interpretation of results.

ICES does not recommend the use of these techniques in routine monitoring assessments under the current state-of-the-art.

ICES recommends that multidimensional techniques, other ordination techniques, and multivariate scaling techniques be further explored for the purpose of producing standard products for use in routine monitoring assessments of biological community data.

Scientific background

The data analysed were time series data on benthic macrofauna from three stations covering the period 1980–2001. Environmental data on nutrients, chlorophyll *a*, and oxygen from the same stations were also obtained from the ICES oceanographic database.

MDS ordinations were made on both abundance and biomass, using untransformed, fourth root transformed, and presence/absence data, thus reducing the influence of the most abundant and highest biomass species on the MDS ordinations.

Biological data and environmental data were then ranked through a similarity matrix, and their rank compared using a Spearman rank correlation coefficient. The correlations are based on the environmental variables giving the best match.

Multivariate scaling techniques are powerful tools for the study of biological communities. An example of the use of such ordination techniques is given in Annex 8.

However, the use of these techniques for monitoring purposes can be questioned. Many ordination techniques such as MDS are well suited for detailed analyses of specific data sets, where local expertise can be used to guide both the data analysis and the interpretation of the results. The challenge is to turn these techniques into tools that can be used in routine monitoring assessments, to reveal correlations between descriptors of biological samples and the level of environmental variables. The use of multidimensional scaling and other ordination and multivariate techniques should be further explored to produce standard data products for use in routine monitoring assessments of biological community data.

More work is needed to understand the relationships between the abundance, biomass, and biodiversity of communities, and the long-term changes in the surrounding environment. These relationships are often complex, and they can be sensitive to particular environmental variables (temperature and the success of reproduction).

10.2 Progress in Understanding the Dynamics of Harmful Algal Blooms

Request

This is part of the 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 algal blooms.

Source of the information presented

The 2003 reports of the ICES-IOC Working Group on Harmful Algal Bloom Dynamics (WGHABD), the Study Group on Modelling of Physical/Biological Interactions (SGBPI), the SCOR-IOC GEOHAB Science Plan, and ACME deliberations.

Summary

Progress has been made in fostering multidisciplinary research on harmful algal bloom dynamics through the establishment of the SCOR-IOC GEOHAB Programme and a new ICES Study Group on Modelling of Physical/Biological Interactions (SGBPI). Simultaneously, the scope of the WGHABD has shifted towards aspects related to harmful species and their impacts. A major step towards closer international collaboration has been taken by organizing an EU-US Workshop on HAB research.

Recommendations

ICES encourages Member Countries to support research initiatives under the SCOR-IOC GEOHAB Programme.

Scientific background

Due to the establishment of the new Study Group on Modelling of Physical/Biological Interactions (SGBPI), and the initiation of the SCOR-IOC GEOHAB Programme, the scope of WGHABD has somewhat shifted, and HAB dynamics are not addressed as extensively as previously. At its 2003 meeting, WGHABD considered historical and retrospective data sets on phycotoxins in shellfish and on phytoplankton, reviewed developments in technologies to detect HAB species, the effect of HABs on survival and fecundity of fish, and methods to detect and quantify phycotoxins in algae and cyanobacteria. One continuing task is to collect and maintain the Harmful Algal Event (HAEDAT) database, and to update the decadal maps of HAB occurrence in the ICES area.

Techniques for the analysis and prediction of the population dynamics of HABs are not well developed and measurements of species-specific growth rates and mortality rates are very difficult. Monitoring is an important aspect of HAB research and WGHABD needs to interact with monitoring programme designs and data interpretation. More environmental data are often needed and sampling should be adapted to local hydrography such as mixed layer depth, circulation patterns, frontal dynamics, etc. Historical data and time series data from sediment and climate studies are important in looking for historical occurrences of HABs.

The Global Ecology and Oceanography of Harmful Algal Blooms (GEOHAB) Programme is now at the stage of implementation. The categories of research include Core, Targeted, and Regional/National studies, as well as Framework activities. The core research includes four projects: 1) HABS in Upwelling; 2) HABs in Semi-confined Eutrophied Zones and Estuaries; 3) HABs in Fjords and Coastal Embayments; and 4) HABs in Stratified Environments. The targeted research includes modelling procedures and approaches, some laboratory experiments, and the influence of water quality on ciguatera development. At present, regional/national research is being prepared by China (CEOHAB), Canada, in the European Atlantic, and in the Baltic Sea.

It was noted that a collaborative agreement has been signed between the European Commission and the U.S. National Science Foundation to foster scientific collaboration. This initiative was subsequently launched by a joint EU-US ECOHAB-EUROHAB Workshop in Trieste, Italy, in September 2002, during which scientists collectively assessed the state of the science, identified gaps in knowledge, and developed a plan for cooperative, comparative studies. This resulted in the proposal of a Joint EU-US Programme on Harmful Algal Blooms, under which comparative research on HAB dynamics in Europe and in the USA will address problems associated with HABs.

11 ENVIRONMENTAL INTERACTIONS OF MARICULTURE

11.1 Implications of the Water Framework Directive in EU Member States on the Sustainability of Mariculture in Coastal and Transitional Waters

Request

This is part of continuing ICES work to keep under review environmental issues relating to mariculture.

Source of the information presented

The 2003 report of the Working Group on Environmental Interactions of Mariculture (WGEIM) and ACME deliberations.

Summary

The EU Water Framework Directive (WFD) was examined by WGEIM as to how it would allow implementation of MarAqua within the context of the European Commission's Strategy for Sustainable Development of European Aquaculture. That strategy clearly builds on reports of the EU MarAqua Concerted Action in the areas of monitoring and aquaculture codes of practice. In addition, it contains elements that reflect the advice that has been offered by ICES on subjects related to integrated coastal zone management. The EU member states' national competent authorities are now in the process of implementing guidelines associated with the WFD. Some aspects of the Directive implementation deserve consideration.

There are still significant uncertainties in the definition of what constitutes a water body and the subsequent classification process. It is not clear whether national authorities will define water bodies of sufficient size to allow for aquaculture to be considered as one of many pressures on water quality within a large body of water. Alternatively, pollution control authorities might call for the definition of much smaller water bodies that allow for a direct relationship between individual significant sources of pollution and much smaller water bodies. The former approach may be considered more desirable and more consistent with the philosophy of the WFD in that it deals with each potential polluter in relation to other activities operating within and impacting on the coastal zone.

In relation to the classification of water bodies, it is not presently clear how the national schemes, and subsequently the intercompared schemes, will accommodate differences in the values of biological or hydrochemical parameters within water bodies. How this is to be done is clearly of importance to mariculture activities, as mariculture sites will present pressures on the environment and some of the parameters of the assessment will be at less than reference status at these sites. Again, this question is not confined to mariculture. Many other anthropogenic activities that result in waste discharges are subject to the same uncertainties.

At this time, it is not clear that member states have given much consideration to the measures required to improve the ecological quality of water bodies (mitigation measures). However, it can be anticipated that additional management and mitigative actions may be required of aquaculture operations in some areas where good ecological status has not been achieved.

It is clear that the promotion of sustainable European aquaculture does not conflict with the requirements of the WFD. Aquaculture should therefore not seek "special treatment". Rather, aquaculture is to be viewed as a coastal zone activity in the same manner as other activities that can influence, and have requirements for, good coastal water quality, such as domestic waste disposal, agriculture, tourism, and forestry. The contribution of aquaculture to ecological conditions assessed as less than good should be considered in the same manner as the contributions from other activities. In developing mitigation schemes, however, it should be noted that aquaculture can also contribute to improving the ecological status of surface water bodies.

The full review of the requirements of the WFD and the implications for mariculture, as provided by WGEIM, is attached as Annex 9.

Recommendations and advice

ICES notes that the conclusions drawn in Annex 9 will be of interest to Member Countries.

ICES encourages Member Countries with mariculture operations in coastal and transitional waters to pay particular attention to the implementation implications of the WFD in light of the above conclusions.

11.2 Potential Impacts of Escaped Non-salmonid Candidates for Aquaculture on Localized Native Stocks

Request

This is part of continuing ICES work to keep under review environmental issues relating to mariculture.

Source of the information presented

The 2003 report of the Working Group on Environmental Interactions of Mariculture (WGEIM) and ACME deliberations.

Summary

Literature was reviewed concerning the impact of several escaped non-salmonid aquaculture species on localized native stocks. The species covered include: sea bass (*Dicentrarchus labrax*), seabream (*Sparus aurata*), cod (*Gadus morhua*), plaice (*Pleuronectes platessa*), haarder (*Mugil so-iuy*), and white sea bass (*Atractoscion nobilis*). Compilation of information on other marine and freshwater species remains ongoing, but most is descriptive life history information.

An attempt was made to identify a series of types of potential interactions that would be subject to risk assessment and potential management strategies. The interactions identified include: competition and trophic interactions, breeding and genetic interactions, and pathogenic interactions. For each of these types of interaction, operational strategies for mitigation are discussed and key areas for research are identified.

Recommendations and advice

ICES recommends that Member Countries conduct research on the interaction between wild and cultured non-salmonid marine fish, which requires both directed field studies and experimentation. Research is also required into the development of cost-effective genetic tools to discriminate between cultured and wild stocks within the same habitat.

ICES recommends that Member Countries document all escape events and report them annually to ICES.

ICES recommends that reproductively sterile fish be used in commercial mariculture operations wherever feasible.

Scientific background

This is a first attempt by WGEIM to collate available published information on the interactions between escaped cultured non-salmonid fish and local native stock. Three examples considered by ACME are provided below.

Sea bass (*Dicentrarchus labrax*)

There is evidence that there are three endemic populations of sea bass (*Dicentrarchus labrax*) in the Atlantic and the Sea of Alboran, the western and the eastern Mediterranean (Patarnello *et al.*, 1993; Cesaroni *et al.*, 1997). This differentiation has been described using allele frequency on six microsatellite loci. The passive retention of larvae on either side of Gibraltar strait was insufficient to explain the persistence of this division. Significant genetic divergence existed between the eastern and western Mediterranean populations (Bahri-Sfar *et al.*, 2000). There were differences within the eastern population as well. Such a differentiation seems to occur in the eastern stock between "groups" of fish that grow in lagoon environments and those that live in the open sea, although both groups appear to share the same breeding areas (Allegrucci *et al.*, 1997; Lemaire *et al.*, 2000). Thirteen enzymatic loci exhibited moderate to high values compared with microsatellites. This was interpreted as evidence that these allozymes are non-neutral. However, only six loci seemed to be implicated in differentiation between marine and lagoon samples, the causes of selection being unknown for the others. A possible scenario of population dynamics of the sea bass between the marine and lagoon habitats has been suggested (Lemaire *et al.*, 2000). Sea bass are known to migrate between coastal and offshore grounds, and a homing behaviour is suspected due to local genetic differentiation over small areas (Allegrucci *et al.*, 1997).

Transfers of eggs and breeders from western populations of sea bass to the eastern Mediterranean have occurred, particularly to Greece and Turkey from France and Italy. Studies on the genetic structure of sea bass in the farms demonstrate a high level of genetic variation in aquaculture stocks. This is due to the continuous replacement of broodstock by individuals originating from the wild, but selective breeding programmes are under way that will inevitably make those stocks genetically distinct from the wild populations.

No information is available on possible interactions between escaped and wild sea bass. This would require directed research into behavioural interactions and the genetic structuring among the wild fish. Nevertheless, the use of western Mediterranean stocks in the eastern Mediterranean is likely to alter local gene frequencies, and individuals from the eastern stocks seem hardly to concentrate in some Greek locations (Divanach, pers. comm.). Wild Egyptian specimens collected in the study by Bahri-Sfar *et al.* (2000) were consistent with the western group, supposedly attributable to the introduction of aquaculture broodstock. Sea bass are reported to mature in the sea cages and, with no containment structure to prevent the dispersion of fertilized eggs, there is a possibility for the release of fry into the wild. However, aquaculture facilities (whether they are land-based units with their effluents or cage systems placed in coastal waters) may be situated in locations where conditions are unfavourable to support the survival of larvae and the escape of cultured specimens to the wild may be irregular and insignificant in numbers to interfere with natural stocks. The behaviour of escaped sea bass has not yet been adequately documented, and it is not known whether these fish would have the fitness to survive under highly competitive conditions. Also, it is not known whether they are capable of migrating to suitable breeding areas, whether maturity cycles are synchronous with those of local fish, and whether interbreeding might occur.

Seabream (*Sparus aurata*)

Allozyme and microsatellite variation and variation in mitochondrial DNA have been examined in seabream. The fish from six different origins from Portugal to

Greece were not distinguishable, indicating substantial gene flow from the eastern Mediterranean to the Azores (Zouros *et al.*, 1998). The comparison with aquaculture stocks demonstrates very low loss of variability among stocks. Nevertheless, the effect of domestication, determined by a measure of the heterozygosity, was apparent in aquaculture stocks. Genetic drift, probably caused by propagation practices, is most likely responsible for the decrease in genetic variation (Palma *et al.*, 2001).

Although farmed stocks outnumber wild fish, the presence of an apparently undifferentiated, single stock of seabream extending throughout the Mediterranean Sea greatly reduces the potential for adverse genetic interactions. The present practice of using wild stock to maintain the artificial broodstock in the hatcheries would lower the possible effects of domestication and inhibit the divergence of aquaculture fish from wild stocks. This would be increased by the maintenance of strains to implement selective breeding programmes.

Wild seabream reproduce in coastal lagoon areas. These are relatively rare environments, and the behaviour of reared fish that could escape is not known. In contrast to sea bass, the risk to fry production in the wild from spawning fish in the aquaculture facilities is not likely to occur, because seabream are protandrous hermaphrodites. Females spawn at three years of age, which is after they are harvested, so only males are reared. Any male contact with females would require (i) that males would have to disperse over a large distance to the breeding areas, or (ii) that mature females would have to be in the vicinity of the cages at the critical time.

Cod (*Gadus morhua*)

There is now evidence that there are at least three reproductively isolated populations of cod within the North Sea and that there may be further isolated populations in the Northwest Atlantic and off Norway. However, the amount of information is limited and research has just begun to investigate this issue. From evidence of Northwest Atlantic stocks, it can be expected that the different reproductive units may intermix to some extent during the summer. However, tagging data showed that there is little exchange between Firth of Clyde cod and those in the Minch, particularly in the North Minch, north of Skye. Cod from the Minch have been caught north of Scotland, but there is little apparent exchange between Minch cod and cod in the Moray Firth.

Whether there will be an impact from escapes from fish farms will depend on the exact nature of the population structure in the wild stock and the genetic nature of the farmed stock of the same species. For instance, it could be envisaged that the impact of escapes would be minimal if farming involved the rearing of wild-caught juveniles from a local stock which was widespread and abundant, and showed a regional rather than local population structure. This would be true even if escapes involved relatively large numbers of fish. On the other hand, a significant impact could occur if the farmed stock were a variety of non-local origin with a narrow genetic base (i.e., a high degree of inbreeding) and it escaped and mixed with a highly structured stock with a local population restricted, for instance, to the fjord with the farm in question. Since cod are now scarce in some inshore waters (e.g., in Scotland), this may increase the impact of escapes of large numbers of non-native farmed stocks.

Jørstad and Nævdal (1992) and Jørstad (1994) reported on an extensive investigation of the effects of mass rearing and release of 0-group cod in fjords and coastal areas of Norway. Each year since 1987, pond-produced cod have been liberated in Masfjorden, a small fjord north of Bergen. The released cod as well as the wild fish and those recaptured in the fjord system have been genetically characterized by electrophoretic analyses of haemoglobin and several enzymes. In 1990 and 1991, about half of the released cod consisted of offspring of broodstock homozygous for a rare allele (Pgi-1(30)). This broodstock was produced by crossing pre-selected heterozygotes for this allele; the homozygotes among the offspring were sorted out on the basis of biopsy sampling of muscle tissue and, when matured, used as parents. Genetic tagging seems to be a useful method both for studying factors controlling survival, growth rate, and dispersal of released cod (Jørstad, 1994), and also for more long-term studies on hybridization and gene introgression between natural and reared populations.

Svåsand (1993) investigated behavioural differences between reared and released cod and wild juvenile cod, using Floy anchor tags and oxytetracycline markers. While differences in individual behaviour patterns occur, no difference in migration patterns were found between wild and reared specimens.

Nordeide and Salvanes (1991) compared the stomach contents and liver weights of reared newly released cod and wild cod; the stomach contents and abundance of potential predators were also described. During the first three days after release, the reared cod fed mainly on non-evasive prey of Gastropoda, Bivalvia, and Actinaria. This is in contrast to wild juvenile cod, which mainly fed on Gobidae, Brachyura, and Mysidacea. Large cod, pollock, and ling preyed upon the released cod immediately after their release, whereas during the months following release the stomach contents of large predators were dominated by Labridae and Salmonidae. The abundance of predators did not seem to increase within the area of release.

Further references relating to cultured and wild cod interactions include Jørstad *et al.* (1994a, 1994b) and Kitada *et al.* (1992). The latter theoretically studied the effectiveness of fish stock enhancement programmes using a two-stage random sampling survey of commercial landings for cod and flounder.

Some mitigation strategies and recommendations for research programmes

To counteract the possible negative effects that escapees (gametes, eggs, juveniles, breeders) from aquaculture operations may have, there is a need to further document numbers of the potential impacts. Blaxter (2000) concluded in his review that "unless a small wild population is swamped by large-scale releases (or stocking) of reared fish, it seems unlikely that the reared fish will out-compete the wild fish". Nevertheless, there seems to be reasonable scientific evidence to take a precautionary approach and to put in place a legislative framework for monitoring fish farm escapes of non-salmonid species, at least until the potential for problems is better understood. There is an urgent need to better regulate the intentional transfers of living animals for commercial purposes. These have impacts on the genetic and pathogenic issues that call for specific mitigative measures.

The behaviour of wild marine fish for which con-specifics have been cultured has not been intensively investigated. There is an urgent need to better understand not only the behaviour of those wild individuals but also the behaviour of released cultured fish when returning to the wild. This would give an assessment of the degree of risk.

The use of reproductively sterile fish is recommended. Currently, triploidy is the only practical large-scale option, given that hormonal treatments are to be forbidden in many countries. In some species, because of their impaired commercial performance (e.g., sea bass), high priority should be given to devising alternative means of sterilization that could be achieved at low cost. Nevertheless, sterile fish may not solve the problem because some sterile individuals have been reported to mimic normal reproductive behaviour and interfere with the natural reproductive activities of their con-specifics.

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12 EFFECTS OF EXTRACTION OF MARINE SAND AND GRAVEL ON MARINE ECOSYSTEMS

Request

This is part of continuing ICES work to review the ecosystem effects of marine aggregate extraction. In 2003, the OSPAR Commission decided that it should make use of the ICES data on sand and gravel extraction in its future assessments, and should not initiate a separate system for collecting similar data.

Source of the information presented

The 2003 report of the Working Group on the Effects of Extraction of Marine Sediments on the Marine Ecosystem (WGEXT) and ACME deliberations.

Summary

An increasing number of ICES Member Countries undertake marine sand and gravel extraction activities and others are looking at the potential for future exploitation. Each year relevant developments on these issues are reviewed and summarized by WGEXT.

Current activities are concerned with developing the understanding necessary to ensure that marine sand and gravel extraction is managed in a sustainable manner.

The extraction of marine aggregates for construction purposes has remained fairly stable over the past few years. Dredging for land reclamation and beach replenishment varied more widely. The largest volume of material extracted is for sand. Significant quantities of material were taken for coastal defense measures.

As in previous years, many of the national reports indicate a great deal of activity on the assessment of the effects of marine aggregate extraction, either as part of the application process for dredging authorizations, or as research.

In 2002, ICES adopted new Guidelines for the Management of Marine Sediment Extraction. These Guidelines were subsequently reviewed by subsidiary bodies of OSPAR in association with their potential adoption for use by OSPAR. This review resulted in several useful proposals for amendments and additions to the Guidelines, which were thereafter considered and adopted by WGEXT at its 2003 meeting (see Section 12.1, below, for details).

The ACME reviewed the revised Guidelines for the Management of Marine Sediment Extraction and adopted them for use in ICES. The final Guidelines are contained in Annex 10.

Recommendations and advice

ICES recommends that Member Countries adopt and implement the 2003 revised version of the ICES Guidelines for the Management of Marine Sediment Extraction, as provided in Annex 10, for use within their countries.

12.1 Revised ICES Guidelines for the Management of Marine Sediment Extraction

The ACME noted that the Guidelines for the Management of Marine Sediment Extraction that had been adopted by ICES in 2002 (ICES, 2003) had subsequently been reviewed by OSPAR subsidiary bodies with a view to their potential adoption for use by OSPAR. The ACME welcomed the very valuable observations made by OSPAR (OSPAR, 2003) on the ICES Guidelines for the Management of Marine Sediment Extraction. These observations emphasized the importance of utilizing an ecosystem approach when evaluating potential extraction operations. It was thus agreed that the Guidelines should be amended to reflect more explicitly the need to:

1) adopt an ecosystem approach to the assessment of the effects and management of marine sediment extraction;
2) recognize in assessments of the potential effects of marine sediment extraction that some ecologically sensitive species and habitats are not subject to specific protection and/or conservation measures under international, European, or national legislation, but nonetheless require special consideration.

The Guidelines were therefore revised accordingly. The ACME adopted this revised version, which is attached as Annex 10.

The ACME reiterated its commitment to the ecosystem approach and noted that the name of WGEXT had been changed several years ago to reflect its changing emphasis away from consideration of environmental and fisheries impacts to the examination of the effects of extraction on the whole marine ecosystem. The ACME also noted that, while the adoption of an ecosystem approach was an important statement of intent, it was not something that could be described in a prescriptive manner. Rather, it embodies a way of thinking about the impacts of human activities that encourages more holistic assessments of impacts on the entire ecosystem.

The ACME also reiterated its support for Strategic Environmental Assessments (SEAs). Reference was made to the EU Directive 2001/42/EC on the assessment of the effects of certain plans and programmes on the environment (the “SEA Directive”), which seeks to provide for a high level of protection of the environment through the integration of environmental considerations into the preparation and adoption of plans and programmes by national, regional, or local authorities which are considered likely to have significant effects on the environment. It was agreed that further development of the approach to SEA in the context of marine sediment extraction was needed.

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12.2 Current Marine Extraction Activities and Results of Assessment of their Environmental Effects

The status of marine extraction and dredging activities is reported annually to WGEXT by participating ICES Member Countries. At present, not all Member Countries prepare an annual report on their activities. However, given that OSPAR agreed in 2003 that this data collection should also serve as its source of data for assessments of sand and gravel extractions and their effects, it is imperative for ICES to consider ways to expand the coverage of the data to all OSPAR Contracting Parties and ensure consistency between data sets submitted by the different countries.

Marine extraction activities

Most of the countries reported that the extraction of marine aggregates for construction purposes had remained fairly stable over the past few years. Data are summarized in Table 12.2.1. Dredging for land reclamation and beach replenishment varied more widely, often reflecting demand from one or two large-scale projects. Significant quantities of material were taken for coastal defense measures. The Netherlands with 16.18×10^6 m^3 had by far the largest extraction activities for beach replenishment, followed by the USA, Denmark, Germany (mainly in the Baltic), the UK, and Poland. Exports are restricted to relatively few countries; the UK with 3.6×10^6 m^3 is the largest, followed by The Netherlands, Poland, and Denmark.

The largest volume of material extracted is sand, with the overwhelming majority coming from The Netherlands. The next largest producer is the UK, but the material is mainly aggregate (sand plus gravel). The proportion of sand and gravel extracted by other countries varies considerably, from 100% aggregate in France to very predominantly sand in the USA.

Extraction figures for the Baltic Sea are not complete, but appear to represent from 5–10% of the total taken from the Northeast Atlantic.

Only two countries, France and Ireland, reported the extraction of maerl, and in Denmark, the extraction of small quantities of glacial till (boulders) was reported.

Review of approaches to environmental impact assessment and related environmental research

As in previous years, many of the national reports indicate a great deal of activity on the assessment of the effects of marine aggregate extraction, either as part of the application process for dredging authorizations, or as research.

Estonia provided information on an Environmental Impact Assessment (EIA) being undertaken in relation to the extraction of 1.3×10^6 m^3 of sand from the Gulf of Finland, for the construction of a new berth in the Port of Muuga. The assessment includes consideration of the impacts on benthic communities, fish, fisheries, seabirds and seals, as well as coastal impacts and impacts on seabed morphology.

France reported the start of a research programme to investigate the impacts of sandpits on bottom morphology in shallow-water areas. The first step has been to test the ability of a state-of-the-art morphodynamic model to reproduce morphological evolutions in a wave tank, and to apply the model to a real site for which morphological evolutions have been monitored over fifteen years.

Germany reported that, until 2001, an EIA was not required for new dredging activities in areas already licensed. It is now obligatory to undertake an EIA if the extraction area exceeds 10 hectares or when the rate of extraction exceeds more than 3,000 tonnes per day. Information was provided on an EIA undertaken in 1999 in relation to the deepening of the Elbe Estuary, and on post-dredging monitoring of the macrobenthos undertaken in 2001 and 2002. The investigation will be completed in 2004. Two research projects were finished in 2002, the results of which will be published shortly. One investigated the regeneration of extraction areas in the North Sea and Baltic Sea. The other assessed the effects of sediment extraction on sensitive macrobenthic species in the southern Baltic Sea.

Information on three EIAs was provided by the Netherlands, covering areas off the coast of South Holland, the Cleaverbank, and for sand extraction for the Westerschelde Container Terminal in the southern part of the North Sea. An update was provided of the PUTMOR study, which is determining the changes in physical parameters inside and outside a large pit situated 10 km off the Dutch coast near Hoek van Holland. The final report is expected in December 2003. Two archaeological maps have been produced. The first provides an indication of the areas with a high chance of finding archaeological and cultural heritage values on the seabed. The second gives the location of archaeological remains, mainly wrecks. A separate project has prepared a map of areas with geomorphological and geological value on the Netherlands Continental Shelf. It is expected to be finalized in 2003.

The UK has published guidance on the issues to be considered as part of an EIA scoping study. Procedural guidance on undertaking benthic surveys was also published. The research into cumulative impacts has been finished and a final technical report is expected later in 2003. Research on the recovery of dredged sites continues, as does work on the role of seabed mapping techniques in environmental monitoring and management. Work on the scoping study for a development plan for marine aggregate dredging continues and will finish shortly. In addition, four new projects have recently started, and will report in March 2004. These will consider:

a) Seabed characterization and the effects of soil structure on the benthos and on benthos recolonization;

b) Impacts of overboard screening on the seabed and associated benthic biological community structure;

c) Preparation of good practice guidance on assessing the impacts of aggregates dredging;

d) Gauging the effects of aggregate extraction on pre-historic deposits on the seabed.

Details of a Regional Environmental Assessment undertaken in the eastern English Channel have been provided. The assessment included consideration at a regional scale of the existing environment, physical impacts, plume effects, marine biology, fish resources, fishing activity, shipping and navigation, and marine archaeology, in addition to potential transboundary effects.

Table 12.2.1. Summary table of national marine aggregate extraction activities in 2002, including EIA activities during 2002.

Country	Aggregate extracted (m^3)	Non-aggregate extracted (m^3)	Aggregate exported (m^3)	Beach replenishment (m^3)	Maps published in 2002	New legislation	EIA initiated	EIA ongoing	EIA finished	EIA published
Belgium	1,620,200	0	0	0	Yes	Yes	Yes	Yes	No	No
Canada	0	0	0	0	No	N/D	No	Yes	No	No
Denmark	5,570,000	2,400	70,000	2,800,000	Yes	Yes	Yes	Yes	Yes	No
Estonia	0	0	0	0	No	No	Yes	No	No	No
Finland	0	0	0	0	No	No	N/D	N/D	N/D	N/D
France	2,427,000	470,000	0	0	Yes	Yes	No	Yes	No	No
Germany	N/D	N/D	N/D	N/D	No	No	No	Yes	Yes	No
Ireland	0	7,700	0	6,300	Yes	No	No	No	No	No
The Netherlands	32,300,000	290,000	2,340,000	16,180,000	Yes	Yes	Yes	Yes	Yes	Yes
Norway	0	115,000	0	0	N/D	N/D	N/D	N/D	N/D	N/D
Poland	532,000	0	167,000	365,000	No	No	No	No	No	No
Sweden	N/D	N/D	N/D	N/D	N/D	N/D	N/D	N/D	N/D	N/D
United Kingdom	12,830,000	0	3,620,000	655,000	Yes	No	Yes	Yes	Yes	Yes
United States	7,180,000	0	0	6,080,000	Yes	No	Yes	Yes	Yes	No

N/D: no data

The United States of America reported on a number of studies, including consideration of regional sediment management, biological monitoring off the Atlantic Coast of New Jersey, and evaluation of the effects of fishing gear on local benthic habitats.

12.3 Application of Risk Assessment Methods as a Tool for Management of Marine Extraction Activities

The ACME reviewed risk assessment methods drawing from material presented in the previous WGEXT report and recent experience of these techniques for evaluating the consequences of marine sediment extraction. It was recognized that a number of marine industries employ formal risk assessment procedures for evaluating the consequences of environmental risk. The most experienced sector in applying risk assessment techniques for evaluating environmental risks in the marine environment was viewed to be the offshore oil and gas industry. It was considered useful to explore the utility (in broad terms) of such approaches and experience for assessing the likelihood of environmental risks arising from marine sediment extraction operations. Hazard assessment is another area where procedures for evaluating risk are well developed, e.g., the Probable Effect Concentration/Probable No-Effect Concentration (PEC/PNEC) risk assessment technique applied in the Netherlands.

The ACME observed that the setting of threshold values or Ecological Quality Objectives (EcoQOs) was one approach for judging the acceptability of environmental risks arising from anthropogenic activities. However, the difficulties of deriving EcoQOs in environments where sediment extraction is ongoing were noted. In particular, it was observed that the derivation of scientifically robust EcoQOs for ecological parameters was problematic, due to the absence of any long time-series data sets for deriving and then testing the behaviour of potential measures. Despite such obvious difficulties, it was noted that there is an increasing number of examples where threshold levels have been set in order to protect the marine environment from the adverse consequences of marine sediment extraction. An illustration of this approach is provided by the Øresund fixed link, where during its construction, targets for vulnerable receptors (e.g., eelgrass and bird species) and overspill material were established. In this instance, monitoring programmes were established to ensure that the agreed threshold levels were not exceeded. In the UK, the sole monitoring programme at Hastings provides another example of a scheme where acceptable limits for a vulnerable receptor were set.

GIS techniques were also suggested as a tool for undertaking spatial and temporal analysis of complex data sets which could be modified to include risk assessment models.

13 POTENTIAL ICES CONTRIBUTIONS TO THE IMPLEMENTATION OF THE OSPAR JOINT ASSESSMENT AND MONITORING PROGRAMME

Request

In 2002, the OSPAR Executive Secretary, on behalf of OSPAR, invited ICES to comment on the revised draft of the OSPAR Joint Assessment and Monitoring Programme (JAMP). In association with this request, ICES has been requested to review the potential ICES contributions to the products listed in the Implementation Table of the OSPAR Joint Assessment and Monitoring Programme.

Source of the information presented

The final draft of the Strategy for the OSPAR Joint Assessment and Monitoring Programme (JAMP); the 2003 reports of the ICES Working Group on Ecosystem Effects of Fishing Activities (WGECO), the Working Group on Statistical Aspects of Environmental Monitoring (WGSAEM), the Marine Chemistry Working Group (MCWG), the Working Group on Marine Sediments in Relation to Pollution (WGMS), the Benthos Ecology Working Group (BEWG), the Working Group on Biological Effects of Contaminants (WGBEC), the Steering Group on Quality Assurance of Biological Measurements in the Northeast Atlantic (SGQAE), the Advisory Committee on Ecosystems (ACE), and ACME deliberations.

Summary

The Implementation Table of the OSPAR Joint Assessment and Monitoring Programme (JAMP) includes more than 75 separate actions to implement, and all but the nine on radioactive substances have some direct linkage to ICES. Some of the entries are themselves composed of a number of different tasks, so the total demands for scientific support are great. ICES considers that the scientific support required to implement the entire JAMP programme far exceeds not just the current science capacity of ICES, but probably also the current science capacity of the countries in the OSPAR area.

Many of the individual areas of activity under the JAMP are areas where ICES has long been active in providing scientific advice and information. For many, in fact, ICES is already established as the primary authority for the provision of scientific support to OSPAR. For the large majority of entries in the Implementation Table, ICES would be the appropriate body to coordinate multinational and interdisciplinary monitoring, data management, analyses, and the development of scientific products, whether it is currently serving that role or not. However, this support will be possible only to the extent that national jurisdictions allocate the scientific resources necessary to deliver the products in the JAMP Implementation Table. To the extent that they do, ICES has the potential to increase the efficiency with which those resources are employed, and ensure the objectivity and rigour of the scientific products developed in support of the OSPAR JAMP Implementation Table.

Recommendations and advice

ICES recommends that OSPAR, as a priority, continue a multi-way dialogue with ICES and national administrations with responsibilities in marine science, with a focus on developing agreements on priorities, a common understanding of roles and responsibilities, and work plans and timetables, for the scientific support to be provided for the achievement of the JAMP Implementation Plan.

Scientific background

The revised JAMP Implementation Table was prepared and released by OSPAR only after most of the relevant ICES Working Groups had held their 2003 meetings. Therefore, the ACME itself integrated the recent products of many ICES Working and Study Groups, in identifying the potential support that ICES could provide for the achievement of the elements of the Implementation Plan.

ICES has approached this task in three steps. For each entry in the Implementation Table, it specified the major types of science programmes and/or information that would have to exist, in order for the particular entry to be achieved. Each of the major programmes or each body of information could itself represent a number of science activities, often requiring substantial effort, coordination, and review. For each major science programme or body of information, ICES then evaluated what potential contribution ICES could make to deliver the necessary science support. Some entries in the Implementation Table are updates of JAMP activities contained in the previous version. For most of these, the nature of both the science support required and the ICES role are already well understood, and can be described in a few phrases. For some of the other entries in the new JAMP Implementation Table, the necessary science support has not been worked out as systematically as in the past. For these components, the narratives in Table 13.1 are more extensive.

ICES stresses that it is reporting on its POTENTIAL contributions. These contributions reflect the expertise that resides in the ICES community of scientists, applied within the established approaches to the work that ICES follows. ICES is confident that its approaches to planning, coordinating, integrating, reviewing, reporting, and developing conclusions and advice from the work of scientific experts are of world-class standard. They increase the efficiency and integration of the work of diverse but separate experts, and add value, rigour, objectivity, and credibility to the integrated scientific efforts and products. However, the degree to which ICES can actually draw on the expertise that resides in the science community of OSPAR and, in fact, the larger ICES community depends completely on the support that national administrations choose to give to the JAMP Implementation Plan. If few experts are actually directed to work on the JAMP Implementation Plan, or are directed to work in isolation from the international coordination efforts that ICES can uniquely provide, then the actual contribution of ICES will be correspondingly less than the *potential* contribution.

In that context, the scientific support required by the entries in the JAMP Implementation Plan, and the potential contributions that ICES can make to the necessary scientific support are tabulated below. Where necessary, a number of comments are added to elaborate or clarify points, or bring out important considerations. This table is not the final word on the role of science – and of ICES – in achieving the components of the revised JAMP Implementation Plan. Rather, it is a comprehensive basis for the three-way dialogue that will be necessary among OSPAR, ICES, and the national laboratories and administrations, as the work plan to deliver the Implementation Plan evolves.

Table 13.1. Tabulation of the general types of scientific support required to achieve the individual entries in the 2003 JAMP Implementation Table, and the potential contributions that ICES could make in providing the required support, realizing that these potentials will require three-way dialogue among OSPAR, ICES, and the cooperating countries of both organizations.

JAMP IMPLEMENTATION PLAN ENTRY		MAJOR SCIENCE SUPPORT NEEDED	POTENTIAL ICES CONTRIBUTION	COMMENTS
AT-1	JAMP Guidelines by 2004 for the selection of data for use in assessments	Expert advice	Provided that the change required to be detected, and that temporal and spatial resolution, etc., are specified, power analyses and the selection of appropriate data series with sufficient power can be carried out by ICES. ICES already provides support of this type to OSPAR, in the context of the previous JAMP arrangements.	Environmental objectives are often stated with no guidance as to how they should be assessed. In order to consider the "relevance" of the suggested purposes AT-1 to AT-3 and to give any useful guidance from a statistical point of view, the objectives need to be quantified, i.e., the magnitude of the changes or differences that needs to be detected, the required resolution, etc., should be specified together with the required power. Furthermore, estimates of variances of the current data that are going to be used are required. If this information is available together with information on the statistical methods that are going to be used and at which significance level, power analyses can be carried out.
AT-2	JAMP Guidelines by 2004 for the assessment of trends in environmental monitoring data	Expert advice	Standard methodologies for assessing time series of marine environmental monitoring data to detect temporal or spatial trends are used routinely by ICES. ICES already provides support of this type to OSPAR, in the context of the previous JAMP arrangements.	
AT-3	JAMP Guidelines by 2006 for determining the frequency of monitoring, and for the selection of monitoring locations in order to secure an adequate geographical coverage, for parameters that are or will be monitored under the JAMP	Expert advice	Provided that sufficient information is supplied (see comments), ICES could estimate sampling frequencies and suggest adequate geographical coverage. ICES already provides support of this type to OSPAR, in the context of the previous JAMP arrangements	Quantitative objectives should be specified, i.e., what is the change required to be detected and what is the spatial resolution needed, what power analyses and spatial autocorrelation are required (see above).
AT-4	Arrangements for data-handling, revised where necessary by 2006 to ensure consistency, efficiency and effectiveness both within OSPAR and with other international organizations	Formal arrangements with explicit expectations and responsibilities specified. Process for periodic review of these arrangements.	ICES has very long experience in data handling, and formal arrangements exist between ICES and OSPAR for the management of some types of data. These arrangements could be expanded by mutual agreement.	

Table 13.1. Continued.

JAMP IMPLEMENTATION PLAN ENTRY		MAJOR SCIENCE SUPPORT NEEDED	POTENTIAL ICES CONTRIBUTION	COMMENTS
AT-5	Methods by 2007 needed to assess the implications of climatic change for the OSPAR maritime area.	Standard sections and/or fixed stations for long-term monitoring of ocean climate. Long-term current measurements for monitoring the strength of the thermohaline circulation. Fully coupled global circulation models (GCM) needed to predict future climate change.	Done routinely by several ICES countries. Competence in WGs to advise on monitoring design and coordination. Competence in WGs to advise on modelling groups that run relevant models.	ICES produces annually a climate status report for the whole North Atlantic. Greatly expanded monitoring efforts are required to provide necessary data for reliable estimates. Specific model run with focus on OSPAR area may be needed.
AA-1	Overview by 2006 of OSPAR Assessment Work 1998–2006	Unclear what level of "overview" is intended, so requirements for science support are difficult to specify.	ICES has had scientific roles in many parts of past OSPAR assessment work, and could serve either a synthesis or a peer review function for this overview.	
AA-2	An assessment in 2010 of the quality status of the OSPAR maritime area and of its sub-regions.	The science support for Quality Status assessments has been specified in past ICES advice, but that advice should be updated well in advance of the 2010 assessment.	ICES is willing and able to contribute to the development of many scientific inputs (data, analytic approaches, etc.), as well as provision of advice on the scientific content of the assessment framework. ICES has conducted peer reviews of past assessments, and is willing and able to conduct peer review of the 2010 assessment.	ICES can either conduct scientific components of the assessment or review assessments conducted by other bodies. However, it would be inappropriate for any group to serve both roles. Depending on who OSPAR wants to participate in the assessments, if the desired teams to conduct the assessments contain many non-specialists, or are too large, it would be more appropriate for ICES to serve the peer review role.
BT-1	Ecological Quality Objectives, initially as part of the pilot project for the North Sea			
1.	Commercial fish stocks above precautionary reference points;	Suitable precautionary reference points for all commercial stocks. Evaluation of status of commercial species relative to reference points. Statistically sound method for combining results of individual stock evaluations into an indicator.	Done routinely at ACFM. Done routinely at ACFM. Under development at present by WGECO.	Part of annual advice to fisheries management agencies. Part of annual advice to fisheries management agencies. Request for this work already received.
2.	No decline in seal population size or pup production of ≥10% over a period of up to 10 years;	Surveys of seal populations and pup production with known accuracy and precision. Robust estimators of seal population size and/or pup production, with sensitivity to detect 10% change.	ICES has expertise and historic leadership role in design and coordination of these surveys. Partially under development. Competence in WGs for complete job.	Some advice already provided on survey design, effort requirements, etc., by WG on Marine Mammal Ecology and statistical experts in various WGs. Greatly expanded survey and monitoring efforts required to provide necessary data for estimators.

Table 13.1. Continued.

JAMP IMPLEMENTATION PLAN ENTRY	MAJOR SCIENCE SUPPORT NEEDED	POTENTIAL ICES CONTRIBUTION	COMMENTS
3. Annual by-catch levels of harbour porpoise should be reduced to levels below 1.7% of the best population estimate;	Surveys of harbour porpoise populations with known accuracy and precision. Robust estimators of harbour porpoise population size and changes. Effective design for by-catch and effort monitoring. Robust analyses of by-catch rates and effort.	Competence in WGs to advise on survey design, necessary effort levels. Competence in WGs to estimate population sizes and trends. Competence in WGs to advise on monitoring programmes. Competence in WGs to estimate and interpret by-catch rates and trends.	ICES has already provided some advice in response to requests from Commissions on theses topics. ICES could provide full scientific support for these scientific tasks. Greatly expanded monitoring efforts are required to provide the necessary data for reliable estimators.
4. The proportion of oiled Common Guillemots should be 10% or less of the total found dead or dying, in all areas of the North Sea;	Sound surveys to quantify occurrences of Common Guillemots being found dead, and presence or absence of oiling. Robust estimator of proportion that are oiled, given the sampling design.	Competence in WGs to advise on survey design, necessary effort levels. Competence in WGs to estimate proportion oiled.	ICES has provided some advice in this area. Greatly expanded monitoring efforts are required to provide necessary data for estimators.
5. No kills in benthic animal species as a result of oxygen deficiency and/or toxic phytoplankton species;	Sound environmental monitoring to detect areas at risk of oxygen deficiency and toxic phytoplankton outbreaks. Selection of appropriate taxa for quantification of mortality. Appropriate surveys and analyses to quantify mortality rates of selected zoobenthos in those areas. Appropriate analyses to establish causal links of mortality events.	ICES can advise on appropriate monitoring designs and necessary effort levels. ICES can evaluate candidate species for quantification. ICES can design, coordinate, and analyse data from targeted surveys, and provide peer review advice on incidence. ICES can review and provide advice from targeted research to establish causality.	Necessity for dialogue with managers on areas and taxa of particular interest, and on desired sensitivity of interpretation of "no kills".
6. A low (<2) level of imposex in female dogwhelks, as measured by the Vas Deferens Sequence Index;	Appropriate surveys to quantify imposex rates of dogwhelks. Appropriate analyses and advice on imposex rates, and changes in occurrence.	ICES can design, coordinate, and analyse data from surveys and provide peer review advice on incidence.	Necessity for dialogue with managers on area of particular interest, for cost-effective programmes.
7. Maximum and mean chlorophyll *a* concentrations during the growing season should remain below elevated levels, defined as concentrations > 50% above the spatial (offshore) and/or historical background concentration;	Appropriate programmes for monitoring chlorophyll *a* throughout growing season. Appropriate methods for quantifying levels and seasonal trends in chlorophyll *a*. Reliable estimates of spatial or background concentrations.	ICES can design, coordinate, and analyse data from surveys and provide peer review advice on levels and trends. ICES can estimate and advise on suitable background and baseline levels and trends. ICES can evaluate status relative to identified background and baseline levels.	The spatial scale on which to work needs to be specified. This has both a scientific and a societal/policy component. ICES can contribute to the former, and participate in dialogue on the latter issue.

Table 13.1. Continued.

JAMP IMPLEMENTATION PLAN ENTRY	MAJOR SCIENCE SUPPORT NEEDED	POTENTIAL ICES CONTRIBUTION	COMMENTS
8. Region/area-specific phytoplankton eutrophication indicator species should remain below respective nuisance and/or toxic elevated levels;	Sound criteria for selecting indicator species. Biological data on candidate indicator species to allow application of selection criteria. Appropriate monitoring programmes for quantifying status and trends of selected indicator species. Sound criteria for specifying nuisance and/or elevated levels of indicator species. Analysis methods to compare monitoring data with specified nuisance and/or elevated levels, with known accuracy and precision.	ICES can select and justify selection criteria for indicator species. ICES can collate and peer review biological data on phytoplankton. ICES can apply selection criteria to develop and advise on suitable indicator species. ICES can design, coordinate, and analyse data from surveys and provide peer review advice on levels and trends. ICES can estimate and advise on suitable background and baseline levels and trends. ICES can evaluate status relative to identified background and baseline levels.	The selection of regions and the desired spatial scale on which to work within regions need to be specified. Each task has both a scientific and a societal/policy component. ICES can contribute to the former, and participate in dialogue on the latter issue.
9. Winter DIN and/or DIP should remain below elevated levels, defined as concentrations > 50% above salinity-related and/or region-specific natural background concentration;	Appropriate monitoring programmes for quantifying status and trends in DIN and DIP. Sound criteria for specifying salinity-related or region-specific natural background concentrations. Analysis methods to compare monitoring data with specified elevated levels, with known accuracy and precision.	ICES can design, coordinate, and analyse data from surveys and provide peer review advice on levels and trends. ICES can estimate and advise on suitable background and baseline concentrations. ICES can evaluate status relative to identified background concentrations.	The selection of regions and the desired spatial scale on which to work within regions need to be specified. Each task has both a scientific and a societal/policy component. ICES can contribute to the former, and participate in dialogue on the latter issue.
10. Oxygen concentration, decreased as an indirect effect of nutrient enrichment, should remain above region-specific oxygen deficiency levels, ranging from 4–6 mg oxygen per liter.	Appropriate monitoring programmes for quantifying status and trends in oxygen concentrations and in nutrients. Sound criteria for specifying region-specific oxygen deficiency levels and background nutrient levels. Analysis methods to compare monitoring data with specified oxygen deficiency levels and background nutrient concentrations, with known accuracy and precision. Appropriate analyses to establish causal links of change in oxygen concentrations to nutrient enrichment.	ICES can design, coordinate, and analyse data from surveys and provide peer review advice on levels and trends. ICES can estimate and advise on suitable background and baseline concentrations. ICES can evaluate status relative to identified background concentrations. ICES can review and provide advice from targeted research to establish causality. ICES can develop operational standards for inferring causal linkages between oxygen and nutrient levels from routine monitoring programmes.	The selection of regions and the desired spatial scale on which to work within regions need to be specified. Each task has both a scientific and a societal/policy component. ICES can contribute to the former, and participate in dialogue on the latter issue.

Table 13.1. Continued.

JAMP IMPLEMENTATION PLAN ENTRY		MAJOR SCIENCE SUPPORT NEEDED	POTENTIAL ICES CONTRIBUTION	COMMENTS
BT-2	Operational EUNIS classification at the level required for mapping and assessment	Consistent classification standards for each EUNIS level. Consistent and operational criteria for assignment of samples to categories within each EUNIS level. Testing of classification system with pilot projects. Specification of spatial scales necessary for mapping and assessment.	ICES has been active in many aspects of development and testing of the EUNIS system, and has advised on its strengths, weaknesses, and potential areas for improvement. Many working groups and different parts of the ICES science community can contribute to further development and implementation of the EUNIS classification system.	There are strong science and policy components to several aspects of the EUNIS system, including the spatial scales necessary for various uses of mapping and various types of assessment. Substantial scientific background work is needed to determine which spatial scales are appropriate for which types of mapping and assessments.
BT-3	JAMP Guidelines for monitoring			
	b. progress towards individual Ecological Quality Objectives (these will be particularly relevant to BM-3(a))	Monitoring requirements are highlighted in BT-1, subpoints 2–10. For each monitoring programme, guidelines will require scientific advice on: • Methods of sampling; • Frequency of sampling; • Spatial allocation of samples; • Laboratory processing of samples (some cases).	ICES has advised on these monitoring aspects many times in the past for various species, populations, and contaminants, and has the competence in WGs to provide the required support and advice for all ten EcoQOs.	Most of the EcoQ elements in the North Sea pilot project will require greatly expanded monitoring efforts, if progress towards achieving the EcoQOs is to be measured with accuracy and precision. Not only does this imply greatly expanded support for field sampling, but also increased support for expertise in experimental and sampling design, statistical analyses of monitoring data, and in data management.
	b. changes over time in species and habitats on the OSPAR List (these will be particularly relevant to BM-3(b))	Special monitoring guidelines are needed for all four points under BT-3a to ensure systematic collection and treatment of information on the abundance and distributions of species and habitats that are to be considered for inclusion on the OSPAR List of Threatened and Declining Species and Habitats. These guidelines will differ from standard monitoring protocols because the species are expected to be uncommon or rare, and/or experiencing rapid change in abundance. Standards for the scientific evaluation of the status of species and habitats relative to the criteria for inclusion on the OSPAR list.	ICES has begun to address this issue in WGECO and other WGs, and is custodian of many of the survey data sets which are the best sources of this information. ICES has the competence in WGs to conduct these evaluations, as well as to critique and advise on the criteria, if requested to do so.	Development of monitoring guidelines for threatened and declining species is likely to require a greater than usual degree of scientific support, including testing in field and computer simulation conditions, because of the special challenges of obtaining the desired accuracy and precision of estimators on species likely to be rarely encountered and distributed in space in ways that may not follow standard sampling distributions.

Table 13.1. Continued.

JAMP IMPLEMENTATION PLAN ENTRY		MAJOR SCIENCE SUPPORT NEEDED	POTENTIAL ICES CONTRIBUTION	COMMENTS
BT-4	JAMP Guidelines for reporting changes in the level and nature of the human activities listed in Appendix 3 (these will be particularly relevant to BM-4)	Guidelines for reporting on the level and nature of human activities require scientific advice on: • Suitable units of effort for reporting the activities; • Standardized methods to scale reported activities to common units of measurement; • Frequency of reporting; • Spatial and temporal scales for reporting; • Sampling strategies for measuring effort within spatio-temporal units.	ICES has undertaken these tasks for sand and gravel extraction, and for mariculture, and is working on methods for quantifying fishing efforts. Many of the other human activities impacting marine ecosystems would require the engagement of social scientists to work with ICES experts, to develop the scientific foundation necessary for progress on this Implementation Plan element.	This Implementation Plan element will require advice from social science as well as biological, physical, and chemical sciences. Ideally they should work together as planning and coordination teams. ICES is just beginning to develop such interdisciplinary working groups.
BM-1	GIS-based habitat maps for the OSPAR maritime area; these could be improved in further information collection, which could start with a one-off survey	Standard survey methodologies to ensure intercompatibility of survey data for marine habitat features. Regularly up-dated and effectively managed data sets, with comprehensive coverage of ocean habitat features and ocean areas. Protocols for incorporating opportunistic data sources into the systematic GIS databases. GIS tools and protocols that are up-dated and revised systematically as the field develops.	ICES can advise on survey design and coordinated aspects of marine habitat surveys. ICES has competence in the Secretariat for data management and administration. ICES has competence in WGs to develop such protocols and advise on their implementation. Several ICES WGs are monitoring developments in GIS tools, and could be a source of technical advice on methods and standards.	ICES and OSPAR are working together on a number of multi-year habitat mapping initiatives. However, GIS is only one of a number of tools featured in those initiatives.
BM-2	Appropriate systems for handling data/information on non-indigenous species	Identification standards and survey protocols that ensure that non-indigenous species are detected and identified correctly when taken in surveys (whatever the main purpose of the survey). Reporting system to ensure that records and biological information on non-indigenous species are collated, analysed, and archived. System for periodic scientific review of reports of non-indigenous species, to detect patterns and trends, and advise on follow-up actions by science and management.	ICES has experience in the design of identification standards and survey protocols, and competence to expand this activity to non-indigenous species. ICES coordinates data reporting and database management for many multi-species, multi-national surveys, and this experience is a basis that could be adapted for ICES to be a repository and data manager for non-indigenous species. ICES WGs are experienced in analysing patterns and trends in marine communities, and would be well placed to conduct such analyses and advise on non-indigenous species.	ICES has been quite active in this area with regard to non-indigenous species in the context of introductions and transfers, and will continue to provide expert advice. The issue of invasive species is a growing priority for Member Countries and clients of ICES, however, it requires increased investment in scientific support to address these growing needs. If this investment is made, traditional ICES expertise, coordination, and advisory roles can all be exercised in this area. This would ensure maximum scientific progress and best advice on follow-up actions, from the various scientific investments that are made.

Table 13.1. Continued.

JAMP IMPLEMENTATION PLAN ENTRY		MAJOR SCIENCE SUPPORT NEEDED	POTENTIAL ICES CONTRIBUTION	COMMENTS
BM-3	Monitoring arrangements for the parameters required to assess a. progress towards achieving agreed Ecological Quality Objectives;	Accompanying the monitoring Guidelines for Ecological Quality Objectives 2–10 (see BT-3a) should be: • coordination mechanisms for conduct of the monitoring and survey programmes; • specified protocols for data reporting and management; • tested methods and tools for statistical analyses of data, with known accuracy and precision. All of these tasks will require scientific advice on methods, and ongoing support for operations.	ICES currently serves all of these functions for data sets on fish, benthos, and oceanographic features. It would be simply further application of existing ICES expertise to provide the scientific support for both selection of methods and coordination of operations.	Individually, the ten elements of the North Sea pilot project make familiar demands on the ICES scientific and advisory roles. Collectively, they represent a very large increase in the magnitude of the scientific demands on the experts in the North Sea area. Major increases in support for science will be necessary for this component of the Implementation Plan to be achieved. However, if this increased investment in science is made, the competence and experience of ICES should give it a central role in the tasks, to maximize the coordinated progress that can be made.
	b. changes in species and types of habitats on the OSPAR List.	Monitoring programmes to quantify the status and trends of species and habitats on the OSPAR list. Statistical methods appropriate for analysing changes in abundance and/or distribution of species that are inherently rare and possibly clustered.	ICES has designed, coordinated, and analysed data from surveys and provided peer review advice on status and trends of some rare or declining species and habitats, and has the competence to conduct or coordinate such activities comprehensively.	Quantifying the status and trends of species that are rare can require significant survey effort using specialized sampling methods and designs. This implies substantial expansion of survey efforts beyond existing surveys targeted at quantifying trends in relatively common (usually exploited) species. Data analysis methods are comparably specialized and demanding.
BM-4	Monitoring/reporting systems for human activities listed in Appendix 3 where assessments under BA-4 have concluded that observation of those activities is justified	Accompanying the monitoring guidelines for human activities (see BT-4) should be: • coordination mechanisms for conduct of the monitoring and survey programmes; • specified protocols for data reporting and management; • tested methods and tools for statistical analyses of data, with known accuracy and precision. All of these tasks will require scientific advice on methods, and ongoing support for operations.	ICES has limited experience in coordinating the monitoring and management of data on human activities in the marine ecosystem, although it has played a coordinating role in relation to aggregate extractions, fisheries, and mariculture. Strong partnerships with social and economic sciences and national authorities would need to be developed, if ICES were to increase its role here.	This Implementation Plan element will require significant cooperation by national authorities to report accurately on human activities – including land-based activities that impact coastal and marine areas – in their national jurisdictions. It will also require the creation and maintenance of a number of new and complex databases on the occurrence of various human activities in space and time.

Table 13.1. Continued.

JAMP IMPLEMENTATION PLAN ENTRY		MAJOR SCIENCE SUPPORT NEEDED	POTENTIAL ICES CONTRIBUTION	COMMENTS
BA-1	Assessment in 2005 of the pilot project on EcoQOs for the North Sea	Conducting assessments of all ten components of the North Sea pilot project assumes that the scientific support specified in BT-1, BT-3a, and BM-3a has all been provided. Conduct of assessments for each of the ten components also requires expert groups, with access to all the necessary data, analytical tools, and competence to conduct peer reviews, draw scientifically sound conclusions, summarize and report on results, and develop advice as needed on follow-up work by science and management.	The potential ICES contributions for these activities are presented in the corresponding sections of this table. ICES has a major role at present in building the scientific and advisory framework for progress on all the elements of the North Sea pilot project, except 6 (imposex in female dogwhelks). ICES would be the logical body to conduct the scientific components of these assessments, even for elements where other groups have had the lead on the development of approaches and methodologies, for two reasons: • ICES experience in conducting inter-disciplinary scientific assessments, integrating the expertise of a particularly wide diversity of experts; • ICES independence, objectivity, and scientific rigour.	Conduct of these assessments will require a major commitment of scientific expertise from the Contracting Parties of OSPAR, regardless of whether or not the assessments are coordinated and overseen by ICES. Using ICES as the arm's length body of scientific experts to conduct the assessments increases efficiency in marshalling scientific expertise, and ensures objectivity and independence.
BA-2	An assessment in 2006 of the status of species and types of habitats that have been placed on the OSPAR List, on the basis of the application of the relevant selection criteria	See BT-3b and BM-3b. Conduct of assessments for each species and habitat on the OSPAR list also requires expert groups, with access to all the necessary data, analytical tools, and competence to conduct peer reviews, draw scientifically sound conclusions, summarize and report on results, and develop advice as needed on follow-up work by science and management.	Comments under BA-1 all apply. ICES has been an active source of scientific advice on the quality of data sources used in developing the OSPAR List. ICES has also conducted evaluations of key components of the overall framework for designating species appropriate for listing.	Comments under BA-1 apply. Some WGs of ICES have expressed concerns about the appropriateness of the criteria being used for listing designations. These assessments would gain value if accompanied by a consideration of the power and sensitivity of the criteria to identify species and habitats which really are at risk of being lost or reduced to very low abundances.

Table 13.1. Continued.

JAMP IMPLEMENTATION PLAN ENTRY		MAJOR SCIENCE SUPPORT NEEDED	POTENTIAL ICES CONTRIBUTION	COMMENTS
BA-3	An assessment in 2006 of the changes in the distribution and abundance of marine species in relation to changes in hydrodynamics and sea temperature	Consistent data sets on abundance and distribution of marine species that extend over periods of at least 1–2 decades. Consistent data sets on sea temperature and hydrodynamic properties that extend over a period of at least 1–2 decades, spatially associated with the biological data sets. Statistical methods and models to relate changes in species abundances and distributions to changes in the oceanographic data. These in turn presuppose knowledge of the appropriate space and time scales for the conduct of such analyses. Knowledge of ocean processes and physical-biological coupling processes, to allow results of the analyses to be interpreted meaningfully.	ICES is the custodian of the major biological and hydrographic data sets needed for the conduct of such analyses. ICES has led the development of such analytical methods and models, and frameworks for interpretation of results, through the work of groups such as the ICES GLOBEC Cod and Climate Change Project, and efforts by several other working groups.	ICES is already the leading scientific body on these integrated tasks, and would be the logical body to be mandated to coordinate the analyses, their peer review and reporting, and development of recommendations or advice based on the results. Dialogue with managers would be necessary to identify marine species of particular interest, given that it would be impossible to assess these changes for every marine species in the OSPAR area, even with the entire scientific capacity of the countries around the North Atlantic.
BA-4	A further assessment in 2009 of the status of the species and habitats that have been placed on the OSPAR List, in the light both of the relevant selection criteria and relevant agreed EcoQOs	See BA-2	See BA-2	See BA-2
BA-5	A series of assessments for the human activities listed in Appendix 3	All the information indicated in BT-4 and BM-4. For each human activity, appropriate assessment methods to evaluate patterns, status, and trends in the data on human activities. To be useful, these would have to include both biological/ecosystem consequences of the activities, and the direct social and economic consequences of the activities.	See BT-4 and BM-4 ICES has significant experience in assessment of biological effects of fishing, at the scale of targeted species and the ecosystem effects of fishing. It would play a central role in this component of the assessment of the effects of fishing. ICES has substantial experience in assessment of biological effects of aggregate extraction and mariculture, and can play some role in those assessments. ICES currently would have little to contribute to the social and economic aspects of these assessments.	These assessments would require close cooperation between the natural science community that is the traditional strength of ICES, and social and economic scientists. These links are being developed in ICES currently, but are no stronger outside of ICES. This work represents a new and challenging demand on the wider European science community.

Table 13.1. Continued.

JAMP IMPLEMENTATION PLAN ENTRY	MAJOR SCIENCE SUPPORT NEEDED	POTENTIAL ICES CONTRIBUTION	COMMENTS
BA-6 A trend analysis in 2007 of all the different human activities listed in Appendix 3 and their collective impact on the OSPAR maritime area	All the information indicated in BA-1, BA-2, BA-3, and BA-5. Methods for scaling trends across different types of human activities, different ecosystem components, and between human activities and ecosystem trends. Methods for partitioning causality of changes in ecosystem components to the various potential natural and anthropogenic forcers.	ICES can play a major role in assembling the diverse natural science expertise needed to support this assessment, in conducting many of the analyses, and in peer reviewing and reporting of results. A major input from social and economic sciences would also be essential, however.	This is exceptionally ambitious. The ICES Symposium on Aarhus II – Changes in the North Sea Ecosystem and Their Causes, attempted a part of this task. Although the symposium was successful at identifying many changes in a variety of ecosystem components, the ability to partition change among alternative causal factors was very limited.
ET-1 JAMP Guidelines by 2004 on monitoring frequency and spatial coverage for nutrients and eutrophication parameters (≡ part of AT-3)	Sound knowledge about the temporal and spatial variability of nutrients and eutrophication parameters. Identification of the critical scales.	ICES can design, coordinate, and analyse data from surveys and provide review/advice on monitoring sampling schemes.	
ET-2 Revision as necessary of existing JAMP Guidelines and development of new Guidelines where they do not already exist, by 2004, to enable monitoring of progress towards individual Ecological Quality Objectives for nutrients and eutrophication effects	Sound knowledge about: • the response of the selected Ecological Quality Objectives to nutrient loading; • the accuracy and limitations of the methods to be used.	ICES can design, coordinate, and analyse data from surveys and provide review/advice. ICES can coordinate QA procedures.	
ET-3 and ET-8 Future development by 2004 and 2006 of the HARP-NUT Guidelines on Harmonised Quantification and Reporting Procedures for Nutrients	Protocols for reporting on nutrient concentrations. Protocols for calibrating and standardizing at-sea monitoring.	ICES has expertise on concentrations and inputs to the sea, and in data management.	Discharges, sources, and losses of inputs have not been a major strength of ICES.
ET-4 Further development where required by 2005 of Ecological Quality Objectives for nutrients and eutrophication effects	See BT-1 points 7–10.	See BT-1 points 7–10.	See BT-1 points 7–10.
ET-5 Further develop harmonized assessment criteria by 2006, together with their respective assessment levels and classification within the Comprehensive Procedure of the Common Procedure	Sound knowledge about the seasonal and spatial variability of nutrients, chlorophyll *a*, selected indicator species, and other eutrophication EcoQOs.	ICES can design, coordinate, and analyse data from surveys and review/advise in setting the assessment criteria.	

Table 13.1. Continued.

JAMP IMPLEMENTATION PLAN ENTRY	MAJOR SCIENCE SUPPORT NEEDED	POTENTIAL ICES CONTRIBUTION	COMMENTS
ET-6 JAMP Guidelines by 2006 for the integrated assessment of emissions, discharges, losses and inputs to, and concentrations and effects in, the marine environment (≡HT-6)	Sound knowledge about the accuracy and limitations of the methods to be used to measure nutrients from different sources. Reliable transport models of nutrients from the emission sources to the marine environment.	ICES can coordinate QA and QC procedures. ICES can review available marine transport models and lead the development/revision of models to be more suitable for this purpose.	
ET-7 An overview by 2006 of predictive models for eutrophication assessment and nutrient-reduction scenarios, including transboundary fluxes within the OSPAR maritime area, and of the possibilities of adopting relevant models for use by Contracting Parties	Development, sensitivity, and reliability testing of ecosystem models.	ICES can review the available ecosystem models and advise on adoption of the relevant models. ICES can review data for model parameterization and advise on strategies to collect new data for model calibration and validation. Competence in WGs to address issues of physical-biological coupling in models.	
EM-1–EM-3 and EM-4 Collection of information in 2004, 2006 and 2008 on emissions to air and atmospheric deposition (via EMEP, EPER)	Appropriate monitoring programme to collect information about nutrient concentrations and deposition to the sea. Specific protocols for collecting and data centre for archiving the data.	ICES has not been active in the areas of emissions to air and atmospheric deposition.	
EM-2 Coherent arrangements by 2004 for the collection of monitoring results required to assess progress towards achieving agreed Ecological Quality Objectives for nutrients and eutrophication effects	Sound methods to produce regionally and temporally integrated data sets on eutrophication EcoQOs.	ICES can advise on selecting suitable integration methods. See BT-1, BT-3, BM-3.	
EA-1 Assessments by 2004 of atmospheric emissions and modelled depositions of nutrients	Appropriate monitoring programme to collect information about nutrient concentrations and deposition to the sea. Reliable atmospheric transport models.	ICES has not been active in the areas of emissions to air and atmospheric deposition.	
EA-2 Assessments by 2005 of temporal trends and (where relevant/possible) spatial distribution for the nutrients where periodic sampling and analysis is undertaken, in particular under CAMP, CEMP and RID.	Knowledge about the anthropogenic impact versus natural variability in nutrient concentrations. Sound criteria for the reference conditions on eutrophication EcoQOs in different subregions.	ICES has already played a significant role in all these scientific activities associated with quantifying the temporal trends in nutrients in marine ecosystems. ICES can advise on setting criteria for reference conditions.	
EA-3 Assessment by 2005 of the pilot project on Ecological Quality Objectives for the North Sea	See BA-1	ICES can review the pilot project report.	

Table 13.1. Continued.

JAMP IMPLEMENTATION PLAN ENTRY		MAJOR SCIENCE SUPPORT NEEDED	POTENTIAL ICES CONTRIBUTION	COMMENTS
EA-4	An assessment in 2006 of the achievement of the 50% reduction target using information obtained through implementation reporting on PARCOM Recommendations 88/2 and 89/4	Sound knowledge on the conditions before the reductions. Reliable methods to measure the nutrient reduction.	ICES can provide data products and expert advice on nutrients in sea water.	The 50% reduction is to be in inputs, where ICES has relatively little activity; however, ICES has substantial expertise in quantifying status and trends in nutrients and chemicals in the sea.
EA-5	Assessment in 2006 of the expected eutrophication status of the OSPAR maritime area following the implementation of agreed measures	Reliable ecosystem models for simulating the impact of reductions in nutrient loading.	See ET-7, EA-2. ICES can review the assessment report.	
EA-6	An assessment in 2007 of the eutrophication status of areas identified under the Common Procedure as problem areas, potential problem areas, and of any non-problem areas where there are grounds for concern	Reliable methods for assessing the eutrophication status and classification of problem areas.	ICES can provide data products, conduct assessments, perform peer review of work by other groups, and provide expert advice.	See comment in AA-2.
HT-1	Guidance by 2003 on a common framework for the establishment of monitoring strategies for each of the substances (or group of substances) on the OSPAR List of Chemicals for Priority Action	Harmonized monitoring of discharges, emissions, and losses of hazardous substances and their concentrations in the marine environment. Sound statistical framework for monitoring programmes. The objectives of the monitoring need to be clearly defined in advance in order that the statistical power of the analysis can be assessed in relation to the aims, and an effective sampling design established.	Analytical methods for priority substances. Validation and QA/QC protocols. Information on concentrations in the marine environment. Advice on statistical and methodological aspects of monitoring strategies.	Done routinely within ICES WGs.
HT-2	Monitoring strategies by 2004 for each of the substances (or group of substances) on the OSPAR List of Chemicals for Priority Action	Harmonized and comparable input data for priority substances in OSPAR countries.	Analytical methods for priority substances. Validation and QA/QC protocols. Advice on data handling/database aspects.	ICES is not routinely involved with assessment of *inputs* of toxic substances, but does have expertise in quantifying status and trends in the sea.

Table 13.1. Continued.

JAMP IMPLEMENTATION PLAN ENTRY		MAJOR SCIENCE SUPPORT NEEDED	POTENTIAL ICES CONTRIBUTION	COMMENTS
HT-3	Assessment tools by 2005 for pursuing the monitoring strategies for substances (or groups of substances) on the OSPAR List of Chemicals for Priority Action. Such assessment tools will include updating of existing background/reference concentrations and ecotoxicological assessment criteria. They will also be consistent with equivalent tools (such as environmental quality standards) developed within the framework of other international organizations and similar systems dealing with the same substances	Surveys of priority substances in the marine environment yielding concentrations with known uncertainty. Framework of BRCs and EACs against which to assess these.	Advice on background concentrations of priority substances in the marine environment. Develop BRCs for priority substances. Expert advice on EACs for priority substances. Review existing EQS and similar national/international guidelines.	ICES can provide inputs to all of these. ICES is planning to co-sponsor with OSPAR an expert workshop to update and extend existing OSPAR background/reference concentrations (BRCs) and ecotoxicological assessment criteria (EACs), which would be a key contribution to this element.
HT-4	JAMP Guidelines by 2006 for pursuing the monitoring strategies for substances (or groups of substances) on the OSPAR List of Chemicals for Priority Action	Guidelines for chemical and biological effects monitoring at source, in rivers, and in the marine environment.	Analytical methods for priority substances. Biological effects monitoring methods for appropriate endpoints. Validation and QA/QC protocols.	ICES can provide inputs to all of these, and is well placed to provide authoritative advice on biological effects monitoring.
HT-5	A review of the JAMP Guidelines by 2006 on biological-effects monitoring and the integration with chemical monitoring	A rationale, framework, and guidelines for undertaking integrated chemical and biological effects monitoring for priority substances.	Analytical methods for priority substances. Biological effects monitoring methods for appropriate endpoints. Validation and QA/QC protocols. Guidelines for the integration of chemical and biological effects monitoring.	ICES can provide inputs to all of these, and is well placed to provide authoritative advice on biological effects monitoring, as illustrated with current advice on integrated biological effects and chemical monitoring. ICES is planning an expert workshop on integrated biological effects and chemical monitoring, which would be a key contribution to this element.
HT-6	Development by 2006 of JAMP Guidelines for the integrated assessment of emissions, discharges, losses and inputs to, and concentrations and effects in, the marine environment (≡ET-6)	Data collected so as to be comparable across OSPAR countries, and a framework for comparing them on a statistically sound basis.	Guidelines for comparing data on concentrations and effects in the marine environment. Advice on statistical aspects.	ICES can advise on aspects of at-sea quantification, but is not routinely involved in work regarding emissions, discharges, losses, and source inputs.
HM-1	A series of information-collection systems by 2005 that implement the monitoring strategies in respect of the priority chemicals (or groups of priority chemicals)	Appropriate electronic reporting and data-handling systems for each priority substance.	Advice on suitable formats and systems for data gathering, storage, and interrogation.	Long experience with ICES databases.

Table 13.1. Continued.

JAMP IMPLEMENTATION PLAN ENTRY	MAJOR SCIENCE SUPPORT NEEDED	POTENTIAL ICES CONTRIBUTION	COMMENTS
HM-2 An information-collection system by 2005 on results of biological-effects monitoring in areas where such effects may occur because of the potential levels of contamination	Appropriate electronic reporting and data-handling systems for biological effects monitoring data.	Advice on suitable formats and systems for data gathering, storage, and interrogation.	Experience within ICES.
HM-3 When appropriate, identification of the likely impacts on the marine environment of substances recorded, inter alia, in source inventories, or identified by screening methods	Information regarding new compounds found to be persistent in the environment and likely to become priority substances in the future, and their potential impacts.	Advice on novel compounds identified during monitoring programmes, source inventories, and screening studies.	Routine role of MCWG.
HA-1 and HA-5 Assessments by 2005 and 2009 of temporal trends and (where relevant/feasible) spatial distribution for the hazardous substances where periodic sampling and analysis is undertaken, in particular under RID, CAMP and CEMP	Comparable and validated data across OSPAR countries and locations of chemical concentrations of priority substances and biological effects, riverine inputs and direct discharges, and atmospheric deposition of contaminants. Suitable statistical methods for identifying and estimating temporal and spatial trends in these parameters.	Assessment guidelines building upon advice on methodology and quality assurance given earlier. Advice on statistical treatment of data. Peer review of assessment.	Experience within ICES.
HA-2 and HA-4 An initial assessment by 2006 and a more elaborated assessment by 2009 of biological effects of hazardous substances in the maritime area	Comparable and validated data across OSPAR countries and locations of chemical concentrations of priority substances and biological effects. Suitable statistical methods for identifying and estimating temporal and spatial trends in these parameters.	Assessment guidelines building upon advice on methodology and quality assurance given earlier. Advice on statistical treatment of data. Peer review of assessment.	Experience within ICES.
HA-3 An assessment every 5 years of emissions, discharges and losses of chemicals identified for priority action. The first assessment will be finalized by 2008	Comparable and validated data across OSPAR countries. A framework and guidelines for assessing the data in relation to the OSPAR strategy.	Advice on statistical data treatment.	Experience within ICES is primarily on the concentrations at sea, and not on emissions, discharges, and losses.
HA-6 A general assessment by 2009 of the development in the quality status of the maritime area that should take into account the results of the assessments under HA-1 and HA-5, HA-2 and HA-4 and HA-3, and the results of any screening of levels of substances in the marine environment covered by HM-3	Integration of all the individual assessments described above so as to describe the quality status of the OSPAR maritime area, and information regarding potential priority substances identified in screening studies and monitoring programmes.	Assessment guidelines, and a review of the environmental occurrence, concentrations, and potential impacts of novel substances. Peer review of assessment. Advice on statistical aspects.	Expertise within WGs.

Table 13.1. Continued.

JAMP IMPLEMENTATION PLAN ENTRY		MAJOR SCIENCE SUPPORT NEEDED	POTENTIAL ICES CONTRIBUTION	COMMENTS
OT-1	Technical annexes to the Guidelines for Monitoring the Environmental Impact of Offshore Oil and Gas Activities by 2004 covering the water column and other issues such as noise if deemed necessary	Knowledge regarding chemical and biological parameters which are of relevance to the monitoring and assessment of the environmental impact of offshore oil and gas activities. Sampling strategies, validated methods of analysis, and associated QA/QC protocols.	Composition and behaviour of drilling fluids and cuttings used in offshore oil and gas development. Knowledge of chemicals used in offshore oil and gas production, and their partitioning behaviour and effects. Composition of produced water/entrained oil and chemicals derived from the formation, and their effects. Effects of noise from seismic surveys and installations (especially floating production, storage, and offloading installations (FPSOs)) on fish and marine mammals. Guidelines for techniques for monitoring biological effects of contaminants in pelagic ecosystems.	Some expertise within WGs, although production chemicals have not been addressed to date.
OT-2	An agreed reference method by 2004 for the determination of dispersed oil content in produced water	Reference analytical method for the determination of dispersed oil in produced water which can be applied routinely on offshore installations.	None – this is almost completed within national regulatory authorities and umbrella organizations such as UKOOA.	
OT-3	A harmonized reporting system by 2004 to compile environmental monitoring data and information related to offshore oil and gas activities	Harmonized methodology and list of required parameters. Agreed reporting format and timetable.	Identify potentially significant impacts of discharged oil and chemicals. Devise/recommend monitoring methods to assess potential impacts. Develop QA/QC protocols. Devise suitable monitoring strategies. Advice on statistical aspects.	Expertise within ICES.
OT-4	Agreed reference method by 2005 for the determination of the content of aromatic hydrocarbons in produced water	Reference analytical method for the determination of aromatic hydrocarbons in produced water. Consider whether alternative methods will also be acceptable and, if so, on what basis.	Advice on suitable GC/MS reference method for the analysis of parent and alkylated aromatic hydrocarbons in produced water, and appropriate validation and QA/QC procedures. Establish performance criteria which will allow the assessment of alternative methods.	Expertise within WGs.

Table 13.1. Continued.

JAMP IMPLEMENTATION PLAN ENTRY		MAJOR SCIENCE SUPPORT NEEDED	POTENTIAL ICES CONTRIBUTION	COMMENTS
OT-5	Assessment tools by 2006 for the impact of hydrocarbons and chemicals from offshore installations	Comparable and valid data for discharged oil and chemicals (formation and production) across the OSPAR area. Framework for risk assessment with which to prioritize the need for controls on the discharge of specific compounds or classes of compounds.	Has been advising on methodology for some years. Statistical input on design of risk assessment procedures. Advice on likely/actual impacts as shown by biological effects monitoring.	ICES has expertise.
OM-1	Information collection through a harmonized reporting system by 2006 to compile environmental monitoring data and information related to offshore oil and gas activities	Harmonized reporting framework, format, and timetable for submission of environmental monitoring data and other information from Contracting Parties. Database in which data can be held, and which can be interrogated to yield data in a suitable format for assessment.	Advice on database design and data handling issues.	Expertise within ICES.
OM-2	Information collection through a harmonized reporting system by 2006 to compile the information collected by Contracting Parties on discharges of hydrocarbons from offshore installations (including major accidental losses of oil) and on chemicals discharged offshore	Harmonized reporting framework, format, and timetable for submission of data on discharges of oil and chemicals from offshore installations from Contracting Parties. Database in which data can be held, and which can be interrogated to yield data in a suitable format for assessment.	Advice on database design and data handling issues.	Expertise within ICES.
OA-1	An assessment by 2007 of the impact on the marine environment of the offshore oil and gas industry	Comparable and valid data on concentrations of chemical and biological parameters in the marine environment in relation to offshore oil and gas installations. Comparable and valid data on discharges and losses of oil and chemicals from offshore installations. A statistical framework for deriving spatial and temporal trends in concentrations of these parameters. Advice on the assessment of the transport and fate of these pollutants.	Advice on the statistical treatment of the monitoring data. Advice on physico-chemical properties and behaviour after discharge of the chemicals, and of their potential impacts.	Expertise within ICES.

Table 13.1. Continued.

JAMP IMPLEMENTATION PLAN ENTRY		MAJOR SCIENCE SUPPORT NEEDED	POTENTIAL ICES CONTRIBUTION	COMMENTS
OA-2	An assessment by 2007 of the possible effects of releases of oil and chemicals from any disturbance of cutting piles	Information regarding the cuttings piles currently in place below offshore platforms due for decommissioning, especially those in which the fields were developed using oil-based drilling fluids, in particular, on the size of the cuttings piles, their composition and current status. Information on the likelihood of disturbance of the piles during removal of the structures (if not removed beforehand), and an assessment of the likely extent and impact of any redistribution of the cuttings pile material. Also to investigate the need for further actions leading to restoration of the marine environment.	Basic physical information has been gathered within a project sponsored by UKOOA, and ICES could assist with advice on physico-chemical properties and behaviour after discharge of the cuttings pile materials, and of their potential impacts. Advice on statistical aspects.	Expertise within WGs
OA-3	An assessment by 2009 of the extent and impact of the offshore oil and gas industry, including the impacts on the marine environment of discharges of hydrocarbons and controlled offshore chemicals, both as they occur and from subsequent remobilization, together with an assessment of the significance for the marine environment of such impacts in relation to the natural changes which are occurring to the OSPAR maritime area	Integration of all the individual assessments described above so as to provide an overall assessment of the extent and impact of the offshore oil and gas industry, and of its significance, in relation to natural changes occurring in the OSPAR maritime area.	Assessment guidelines, and advice regarding the progress of natural change. Advice on statistical aspects.	Expertise within WGs.

14 DATA HANDLING

14.1 Handling of Environmental Data: Data on Contaminants in Marine Media, Biological Effects, Fish Disease, and Biological Community Data

Request

Item 4 of the 2003 Work Programme from the OSPAR Commission: to carry out data handling activities relating to:

4(a) contaminant concentrations in biota and sediments;

4(b) measurements of biological effects;

4(d) data on phytobenthos, zoobenthos and phytoplankton species.

Item 5 of the 2003 Work Programme from the OSPAR Commission: to continue with the development of a relational database for data on contaminants in biota, sediments and water. This activity should cover the following issues:

Conversion to relational database;

Conversion of screening program;

User-friendly data and information import/export facilities;

Development of programs for data products, including web inventories.

Contract from the Helsinki Commission (HELCOM) to serve as the 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, and extended for a second three-year period.

Contract from the Arctic Monitoring and Assessment Programme (AMAP) to serve as Thematic Data Centre for the marine component from 1998–1999, extended to 2000–2001.

Source of the information presented

Progress report from the ICES Marine Data Centre and ACME deliberations.

14.1.1 Database holdings and data submissions

The environmental data held by the ICES Marine Data Centre include the following types:

1) contaminants in marine invertebrates, fish, birds, and mammals (approximately 515,000 records);
2) contaminants in sea water (approximately 330,000 records);
3) contaminants in marine sediments (approximately 109,000 records);
4) biological effects of contaminants (approximately 4,000 records);
5) fish disease prevalence (approximately 800,000 records);
6) biological communities (phytoplankton, zooplankton, phytobenthos, and zoobenthos) (only a few submissions have been received so far).

The spatial and temporal coverage of data remains lower than expected based on commitments to OSPAR, HELCOM, and AMAP. The status of submissions of environmental data to the ICES Marine Data Centre can be found on the website at: http://www.ices.dk/env/index.htm.

14.1.2 Problems related to the submission of data

The relatively low number of data submissions to the ICES environmental database is likely to be caused by several factors; some of these have been emphasized earlier by data submitters and the ICES Data Centre. In summary, the main obstacles to the submission of data to the ICES environmental database can be categorized as follows:

1) The ICES Environmental Data Reporting Format itself. The fixed-file format has been mentioned as a problem for several laboratories since it requires more programming to transfer data from national databases to this format than if the format were more flexible or even free (as for the oceanographic data).

2) Data not collected in accordance with the guidelines for sampling. The sampling methods are not always in accordance with those prescribed in the monitoring guidelines, which might conflict with the data structure of the database developed. This has been the case for the biological community database in some situations.

3) Lack of priority in the national laboratories. It is clearly the impression of the ICES Data Centre that the priority put on the transfer of data from the national databases to the ICES Data Centre is low in many countries. The low priority is not necessarily due to any lack of interest in submitting data, but could be due to a lack of resources.

A solution concerning the reporting format could be to accept data in any kind of format; however, this would put a much higher demand on the capacity within the ICES Data Centre concerning the transfer of data into the database. This capacity is not available at present. A more likely solution would probably be to develop a more flexible format (e.g., XML). The lack of compliance with the monitoring guidelines must be solved within the Commissions which issue these guidelines (HELCOM and OSPAR). Finally, the low priority on data transfer is not easy to solve when there is no legal enforcement for the submission of data within HELCOM and OSPAR. However, it should be emphasized that the allocation of resources for the handling of data is often not in proportion to the resources allocated for the collection and analyses of the actual samples.

14.1.3 Major data products and requests

Indicator reports on temporal trends of contaminant concentrations in herring from the Baltic Sea and temporal trends of nutrients and chlorophyll *a* in sea water have been prepared as part of the contracted work for HELCOM. The development of indicator reports within the HELCOM framework is intended to be an annual event. Both indicator reports are based on data in the ICES databases. The data on chlorophyll were supplemented with data from the German data centre in order to include the most recent data. Temporal trends were tested using the non-parametric Mann-Kendall test. An example of the data products contained in these reports is shown in Figure 14.1.3.1.

As a part of the work within the ICES Working Group on Statistical Aspects of Environmental Monitoring (WGSAEM), data on benthic macrofauna have been analysed using non-parametric Multidimensional Scaling (MDS) and other techniques. Time series on benthic macrofauna from three stations covering the period 1980–2001 were used. The data were obtained from the old HELCOM database stored at ICES and from the National Environmental Research Institute, Denmark. To complement the benthic macrofauna data, environmental data on nutrients, chlorophyll *a*, and oxygen from the same stations were obtained from the ICES oceanographic database. The results of these analyses are discussed briefly in Section 10.1.2, above, with the details in Annex 8. It is recommended that multidimensional scaling and other ordination and multivariate techniques should be further explored to produce standard data products for use in routine monitoring assessments of biological community data.

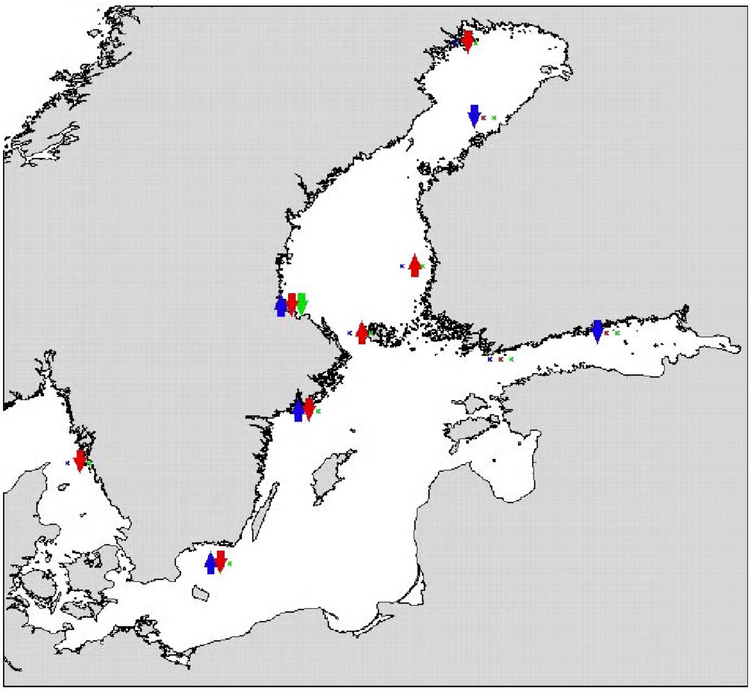


Figure 14.1.3.1. Temporal trends in trace metals: cadmium (blue), lead (red), and mercury (green) during the period 1980–2001 from several areas in the Baltic Sea based on the non-parametric Mann-Kendall test. The symbols are: X = no significant trend (P>0.05); ⇑ = significant upward trend (P< 0.05); ⇓ = significant downward trend (P< 0.05).

The ICES Marine Data Centre has provided data to the European Environment Agency (EEA) based on a requested access to raw and integrated data collected in OSPAR and HELCOM monitoring programmes from 1985 to the present. This request is an annual update of an earlier request. The data sets handled were: 1) harmful substances in biota; 2) harmful substances in sediment; and 3) eutrophication-related parameters in water. These data are intended to be used to prepare indicator fact sheets, and all the raw data utilized in the derivation of the indicators will be published by the EEA Water Topic Centre.

The test version of a HELCOM inventory presented last year (ICES, 2003) will be updated and further elaborated to cover the needs of the HELCOM community. This test version can been found on the web page http://www.ices.dk/ocean/asp/helcom/helcom.asp. The inventory is being developed as a collaborative effort between the Danish National Environmental Research Institute and the ICES Marine Data Centre.

14.1.4 Future development of the environmental databases

Limitations in the present structure of the databases held in ICES and the growing request to integrate information across several compartments of the ecosystem have led to the initiation of the development of a new data structure that integrates several of the present databases held in ICES. The process of integrating data from oceanography (bottle data) and the environmental databases (contaminants in biota, sea water, and sediments, biological effects, fish disease, and biological community data) has progressed and the new data structures are under consideration. Apart from integrating these various data types, it will also include the new data types on biological effects of contaminants.

The task of integrating these various data types into one database has not proved to be easy due to several circumstances. The definition of fields for the reporting of biological effects has not been particularly clear, primarily due to a lack of feedback from the various specialists in this field. Furthermore, the large amount of information required on methods and quality assurance complicates the data structure. In general, there seems to be a need for developing a generally agreed level of what should be stored in the database with regard to metadata.

Recommendations and advice

ICES recommends that Member Countries urge their national laboratories to submit data according to their commitments to the marine Conventions. The submissions of data are crucial for the ability to make assessments of the status of the marine environment. ICES emphasizes that the future integrated ICES database will constitute a powerful tool facilitating more holistic assessments that integrate data from different compartments of the ecosystem. However, the successful use of this tool is highly dependent on a good spatial and temporal coverage of the data.

Reference

ICES. 2003. Report of the ICES Advisory Committee on the Marine Environment, 2002. ICES Cooperative Research Report, 256: 101–104.

14.2 Handling of Nutrients Data for the OSPAR Commission

Request

Item 4(c) of the 2003 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

Report from the ICES Oceanographer and ACME deliberations.

Summary

Tables 14.2.1 and 14.2.2 provide information on the number of nutrient stations in the ICES database for the past decade from a wide area of the Northeast Atlantic (30°–80°N; 20°W–12°E). Table 14.2.1 lists data received from all sources and all projects. Table 14.2.2 lists only those stations which have been clearly identified as OSPAR nutrient monitoring data by the data providers. These are usually surface-only data. These tables do not take into account data received, but not yet processed, by the Secretariat. There is now a significant amount of these data, and there are various reasons for the delay in handling them. Notwithstanding this, the long-term delay in the provision of data from a number of Norwegian sources and, increasingly, the Netherlands is a significant problem delaying the analysis of these data to support the OSPAR Comprehensive Procedure of the Common Procedure for the Identification of the Eutrophication Status of the OSPAR Maritime Area.

Figure 14.2.1 shows the distribution of nutrient stations in part of the Northeast Atlantic during the first three months of the year. It is this period of the year that features in most of the data products required by OSPAR. 7,669 of the 25,404 stations were sampled at that time of the year, indicating that there is no extra sampling effort at that time, in spite of the relevance of such data to the analysis of the eutrophication status in the area.

Table 14.2.1. Number of nutrient stations in the ICES database, by year and country during the past decade (as of June 2003).

Country	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	Total
Belgium	169	129	136	118	227	145	142	115	0	0	0	1181
Denmark	379	381	134	200	249	300	230	220	221	174	76	2564
Finland	0	0	4	0	28	0	0	3	0	0	0	35
France	7	10	114	7	52	10	0	0	0	0	0	200
Germany	905	369	381	355	454	559	116	22	0	0	0	3161
Iceland	86	162	0	8	50	69	77	0	50	0	0	502
Ireland	50	0	0	0	0	0	0	0	287	251	0	588
Netherlands	979	904	987	566	594	97	49	49	64	20	0	4309
Norway	820	966	472	404	515	391	234	281	0	0	0	4083
Poland	19	0	0	86	272	0	0	0	0	0	0	377
Russia	11	0	0	0	0	0	0	0	0	0	0	11
Spain	118	133	43	2	0	0	0	0	0	0	0	296
Sweden	293	417	265	229	220	199	113	107	153	78	0	2074
UK	871	487	566	769	535	945	488	434	603	221	51	5970
USA	0	0	0	0	53	0	0	0	0	0	0	53

Table 14.2.2. Number of OSPAR-specific nutrient stations in the ICES database, by year and country during the past decade (as of June 2003).

Country	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	Total
Belgium	133	127	113	118	201	115	123	115	0	0	1045
Germany	269	0	164	129	131	175	34	22	0	0	924
Ireland	0	0	0	0	0	0	0	0	287	251	538
Netherlands	921	861	879	465	547	0	0	0	0	0	3673
UK	0	0	0	0	0	221	311	152	0	0	684

14.3 Advice and Standard Data Products for Devloping the OSPAR Common Procedure for Identification of the Eutrophication Status of the Maritime Area

Request

Item 2.1 of the 2002 Work Programme from the OSPAR Commission: to provide assistance in the preparation of data products based on the relevant data series available in ICES databanks, for inclusion in an assessment report of the eutrophication status of the OSPAR maritime area.

Source of the information presented

Report from the ICES Oceanographer and ACME deliberations.

Summary

No new products have been prepared for OSPAR in support of the Comprehensive Procedure of the Common Procedure for the Identification of the Eutrophication Status of the OSPAR Maritime Area (COMPP), the reasons for which are explained in Section 14.2, above. In addition, because of the advanced stage reached in this evaluation, which is to be presented to the 2003 OSPAR Ministerial Meeting, no new products are required for this particular purpose.

The 2002 ACME report on this item (ICES, 2003) highlighted the need to move towards the evaluation of nutrient budgets and a proposal was made for a greater involvement of the LOICZ (Land-Ocean Interactions in the Coastal Zone) community. This involvement has not yet occurred, mainly because the Theme Session set up

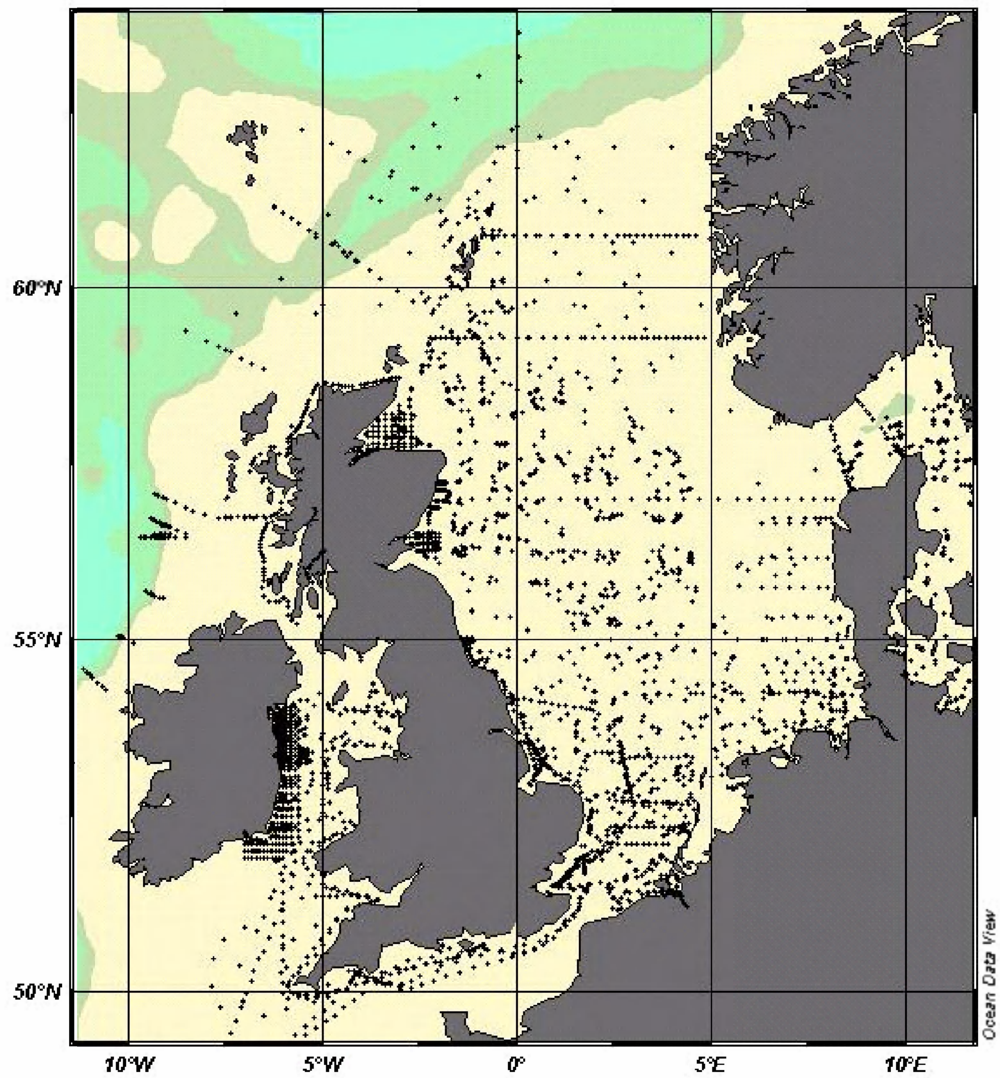


Figure 14.2.1. Distribution of January–March nutrient stations in part of the Northeast Atlantic for the period from 1993–2003.

to address this issue at the 2003 ICES Annual Science Conference has attracted only three papers. One of these is from the Study Group on Modelling of Physical/Biological Interactions community, which also plans to address this issue under one of its terms of reference. The Study Group will assess how an ecosystem will respond to nutrient load reductions by conducting a joint review of their experimental simulations of nutrient load reductions in the Baltic Sea, the North Sea, and the Skagerrak.

The main objective of COMPP was to produce a harmonized assessment of the eutrophication status of the OSPAR maritime area. However, the last meeting of the OSPAR Eutrophication Committee (EUC) which was attended by the ICES Oceanographer demonstrated that this has become an elusive objective, as significant disharmony had set in due to different interpretations of the steps of the procedure by various OSPAR Contracting Parties. This disharmony has its roots in a problem foreseen by the Advisory Committee on Ecosystems (ACE), who noted in 2002 that "When several EcoQ elements can only be considered as an integrated set, there will have to be a second set of rules for how the status of the ecosystem on each EcoQ element is combined into that integrated set, to provide a single clear message on management action. These rules may prove challenging to develop" (ICES, 2002). As COMPP requires the integration of five EcoQ elements, and as the rules for integrating them have not been developed, this has allowed subjectivity to creep into the assessment procedure, resulting in differing interpretations from country to country. The EUC meeting was unable to resolve these differences, which has resulted in a final product which lacks harmony. It was acknowledged, however, that this was only the first attempt to produce a eutrophication assessment and these difficulties would have to be addressed in order to guard against repeating them in subsequent applications of COMPP.

Recommendations and advice

ICES notes that OSPAR has scheduled its next thematic assessment for eutrophication for 2006, and that the ICES databases on nutrients, oxygen concentrations, and other supporting parameters will be primary sources of data for that assessment. ICES is concerned that, if the current poor provision of these data to ICES continues, the thematic assessment will not be possible. ICES notes that for some countries data flow is excellent, and hopes that such provision of data is maintained. However, in order to allow the thematic assessment to meet reasonable scientific standards for completeness and rigour, ICES urges Member Countries to continue to submit these data in a timely manner and in all cases by no more than one year after collection. In light of historic non-compliance of many Member Countries with commitments to contribute these data to ICES according to agreed forms and timetables, ICES may require assistance from OSPAR to bring forth such submissions.

References

ICES. 2002. Report of the ICES Advisory Committee on Ecosystems, 2002. ICES Cooperative Research Report, 254: 55–56.

ICES. 2003. Report of the ICES Advisory Committee on the Marine Environment, 2002. ICES Cooperative Research Report, 256: 99–101.

14.4 ICES Phytoplankton Checklist

Request

This issue is of importance to the development and maintenance of databases for biological communities.

Source of the information presented

The 2003 report of the Working Group on Phytoplankton Ecology (WGPE), comments from the ICES Marine Data Centre, and ACME deliberations

Summary

The ACME noted that the status of the ICES/IOC Checklist of Phytoplankton and other Protists is unclear. However, it is hoped that a compiled list of phytoplankton species will be finalized by October 2003 based on work carried out intersessionally within the WGPE.

The ACME noted that some questions have been raised concerning the decision of ICES to use the Integrated Taxonomic Information System (ITIS) as a standard for use in data submissions. In particular, the WGPE has expressed concern about using the ITIS coding system. The main concern was that ITIS contained a very small number of phytoplankton species from the European marine waters and that there appeared to be a lack of maintenance of the system.

The ACME emphasized that the ITIS system is in fact maintained and is based on the submission of information from the taxonomic experts on the various taxonomic groups and geographical areas. Contacts to ITIS made by the ICES-IOC Working Group on Harmful Algal Bloom Dynamics (WGHABD) confirm that the focus has not been on phytoplankton species and that the geographical area is largely North American for most taxonomic groups. However, ITIS is open to collaborators and users and, being a member of the Global Biodiversity Information Facility (GBIF), ITIS has agreed to adhere to evolving community standards.

The ITIS list is admittedly not comprehensive for all taxonomic groups and geographical areas. However, the other taxonomic code systems, e.g., RUBIN, are either no longer maintained, or suffer even more from a lack of taxonomic and geographical coverage. The submission of data based on a range of different species code lists will make the extraction and use of the biological community data a very tricky and cumbersome exercise.

The maintenance of taxonomic lists is not an easy task and the amount of taxonomic expertise that has to be consulted is huge. Therefore, the ICES Marine Data Centre cannot accommodate this task with the present resources.

Recommendations and advice

ICES recommends that the list of phytoplankton species compiled by WGPE should be submitted to ITIS with the aim of making a more complete code list of species from European marine waters.

ICES further recommends that taxonomic lists for other groups of species, in the possession of various institutions, national laboratories, experts, etc., should be submitted to ITIS.

ANNEX 1: AN OVERVIEW OF THE PROGRAMME OF THE ICES WORKSHOP ON BIOLOGICAL EFFECTS OF CONTAMINANTS IN PELAGIC ECOSYSTEMS (BECPELAG)

Project aims:

- To improve knowledge about effects of pollutants in the pelagic zone
- Pelagic zone
  - i) is the place where the most sensitive life stages are found
  - ii) is affected by discharges from rivers and offshore industry
- Propose (set of) bio-effects methods for monitoring
- Use project to assess various biological effects methods

Working aims:

- Look for relations between contaminants and effects
- Try to account for interfering quantities (e.g., temperature, etc.)
- Identify biomarkers with response
- Identify biomarkers with similar / non-similar behaviour
- Response + unique behaviour qualify for a test battery
- Non-response or similarity with others demand further investigation

Geographical regions:

- Statfjord transect:
  - Four stations on a line away from the Statfjord oil field (codes S1–S4)
  - Selected because of pollution gradient from S1 down to S4
  - S4 should be nearly unaffected by oil rigs
- German Bight transect:
  - Four stations on a line from the Elbe estuary to the central North Sea (codes GB1–GB4)
  - Selected because of (presumed) pollution gradient from GB1 down to GB4
  - GB4 should be nearly unaffected by river inputs

Data compartments:

- pelagic environment
- field-collected organisms
- caged organisms
- bioassays

Data compartments: pelagic environment

- Hydrography
- General pollutant load
- *In situ* measurements of PAHs in
  - water
  - biota
  - SPMDs (semi-permeable membrane devices)
  - DGTs (diffusive gradient in thin film)
- Brominated compounds in pelagic organisms

Data compartments: field-collected organisms
(herring, saithe, cod, dab, mackerel, plankton, fish embryos, blue mussels)

- Biomarkers: EROD, CYP1A, AChE, CEA, antioxidant responses
- Fish embryo aberrations
- Virus prevalence
- Liver histopathology

Data compartments: caged organisms
(cod, stickleback, blue mussels)

- Biomarkers: EROD, CYP1A, AChE, BaPH, GST, CEA, MT, DNA adducts, vitellogenin, scope for growth, lysosomal stability
- PAH metabolites
- histopathology

Data compartments: bioassays
Fish cell lines, liver tissue, fish eggs
Water extracts, UV light, bile extract

- Biomarkers: AChE, Cytoxicity (various forms), reproduction capability

Sampling schedule:

- Samples were taken between March and September 2001
- Not all measurements were mentioned so far
- Not all mentioned measurements at all times and stations
- Material availability crucial criterion!
- Caged organisms deployed in April/May, collected in June (about 8 weeks of exposure)
- Eight cages (four Statfjord, four German Bight; one lost in German Bight)
- 25 cod per cage (80% survivors)
- No surviving stickleback
- No SPMD, DGT at one Statfjord station

Project organization:

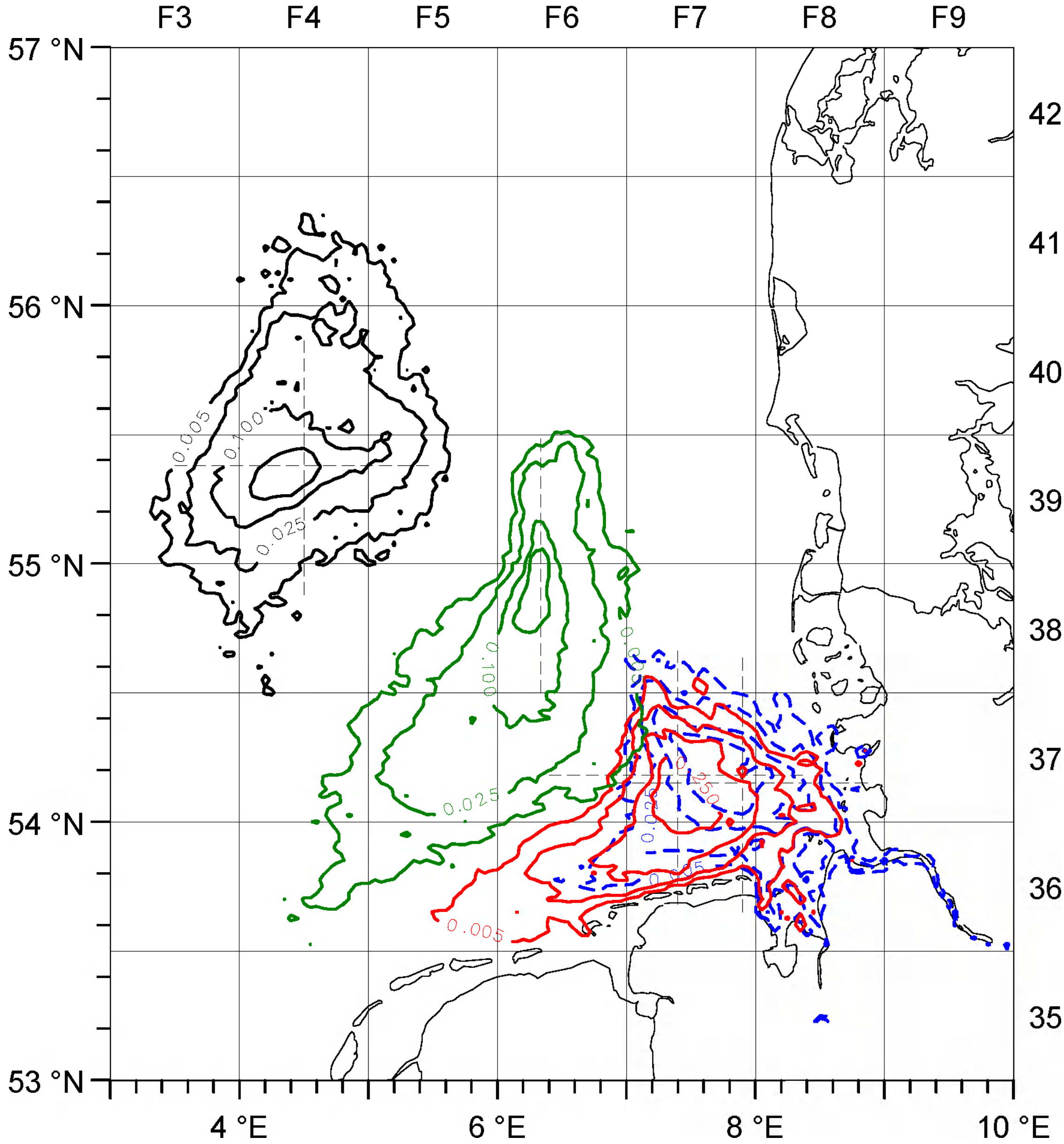
- About 30 research groups from Europe
- Five research vessels, seven cruises
- Project originator: Ketil Hylland, NIVA, Oslo
- Project steering committee:
  - Ketil Hylland (N), Björn Serigstad (N), Jarle Klungsoyr (N), Dick Vethaak (NL), John Thain (UK), Kevin Thomas (UK), Alisdair McIntosh (UK), Gerd Becker (D), Thomas Lang (D), Werner Wosniok (D)

Funding:

- Norwegian Environmental Agency
- ICES
- Results to be published in SETAC series

Data examples (1)

Simulated spatial distribution of the origin of particles arriving at sampling stations GB1–GB4 between 22 March 2001 and 20 June 2001. Numbers along contour lines denote the percentage of particles originating in the interior of the contour. Contour levels are 0.005, 0.025, 0.1 and 0.25%, uniformly for all stations. Dashed contours refer to GB1.



Simulation done with the Operational Model of the BSH, Hamburg

Statistics (1)

- Standard methods for data records referring to the same individual:
- Linear, generalized linear model
- What if

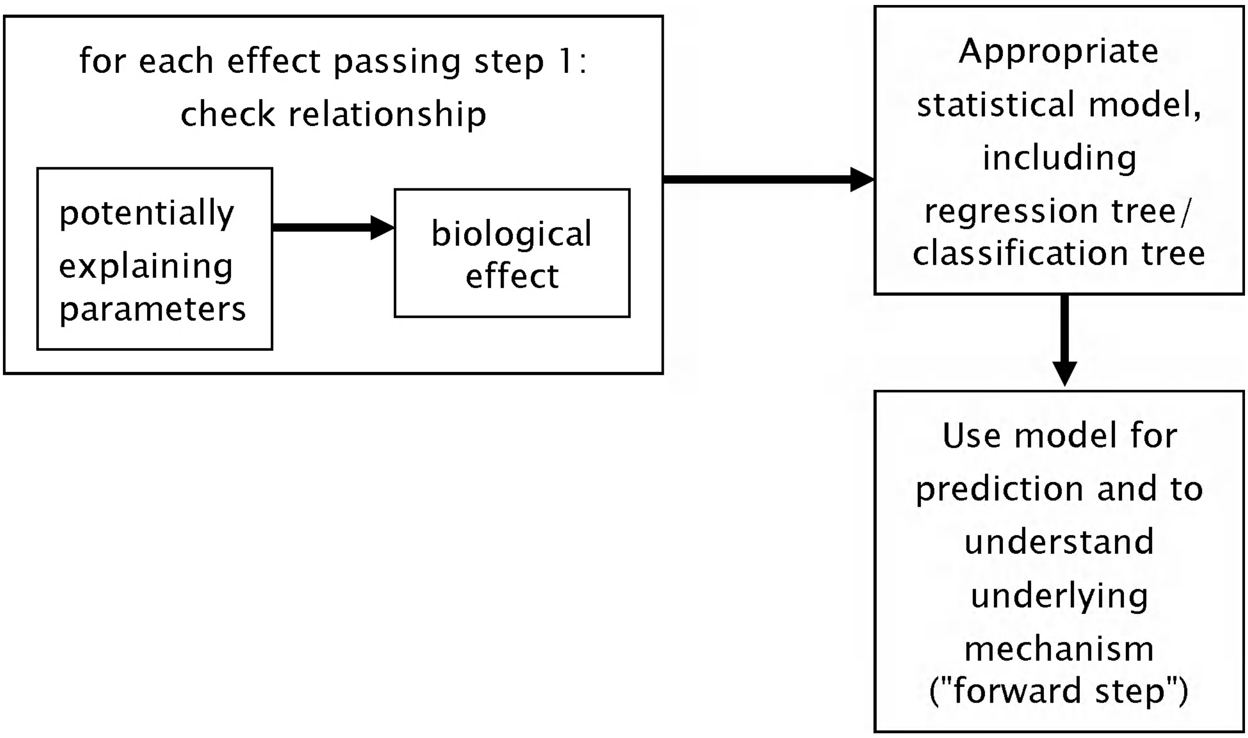
x measured with error
likely to be distributed over area
referring to condition outside the individual

Statistics (2)

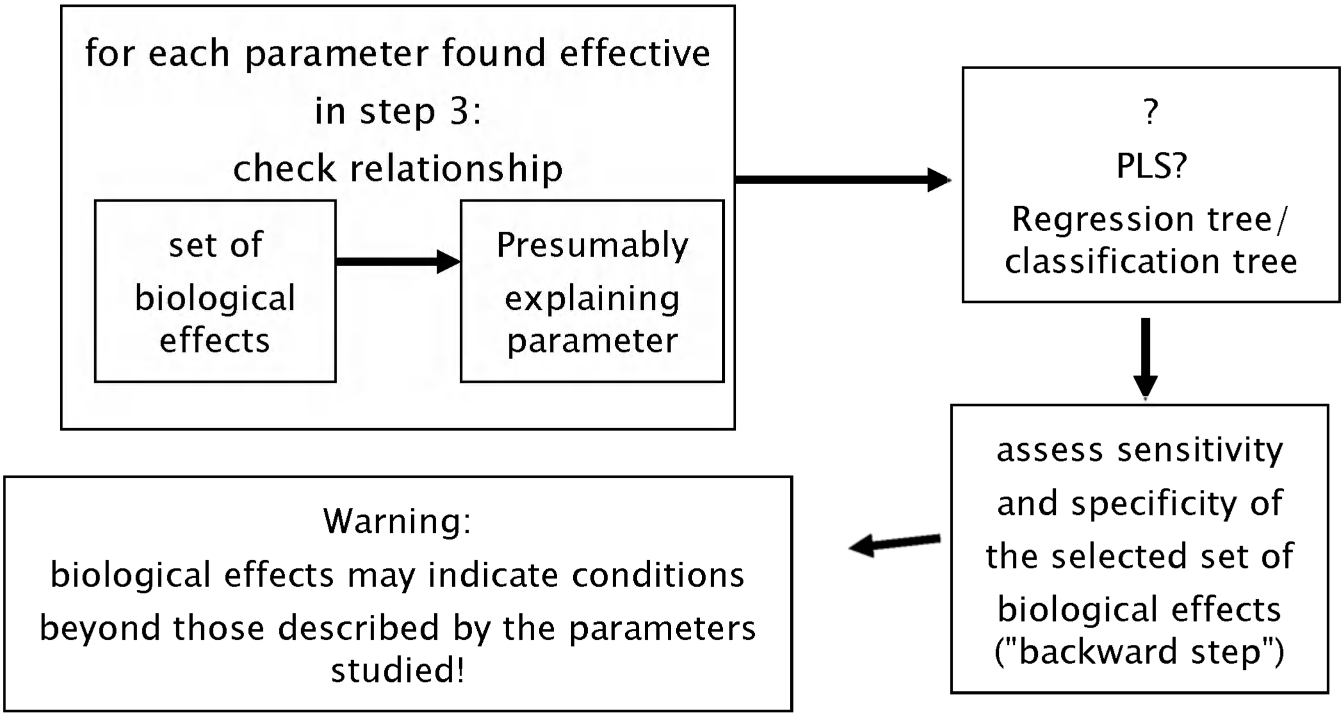
y = f(*x*),
x likely to be distributed over area
not measured within individual

- Ignore variation, take means?
  - OK for orientation, bad if dispersion of *X* is large
- Mixed effects model?
  - Requires estimate at least for standard deviation (SD) of *x* – from 3 repeats?
  - "Educated guess" for SD, then mixed model? Simulation?

Step 3 of the integrated assessment



Step 4 of the integrated assessment



ANNEX 2: INVENTORY OF NATIONAL PROGRAMMES FOR TREND MONITORING OF CONTAMINANTS IN SEDIMENTS

The theory behind the use of sediments as a tool in environmental monitoring is the knowledge that the finer particles in the sediment originate from the suspended particulate matter, and that these particles are carriers of contaminants associated with them. Sediments often act as a sink for contaminants, with older deposits being progressively buried by new deposition. Analysis of stable sediment cores can, therefore, provide information on changes in contaminant fluxes to the sediment over time. If necessary, differences in the sediment matrix can be normalized between different samples, either by recalculation methods or by selecting/transferring the samples to comparable composition.

Definition of temporal trend monitoring

Temporal trend monitoring may be described as repeated measurement or study of a certain parameter in a certain medium at selected time intervals and using constant or demonstrably comparable procedures. In this context, the medium is marine bottom sediments and the parameters are anthropogenic contaminants.

Objective of temporal trend monitoring

The objective of temporal trend monitoring is to identify temporal changes of contaminant concentrations in sediments.

Temporal trend monitoring of contaminants in sediments can, in principle, be executed by the following techniques:

1) Retrospective studies, i.e., studies of down-core concentrations;
2) Comparing concentrations from repeated surface sediment sampling at the same sites;
3) Comparing concentrations of repeated surface sediment sampling at different sites but within the same area;
4) Comparing concentrations in suspended particulate matter (SPM) collected regularly by sediment traps at the same sites.

Advantages/disadvantages of the different techniques

Studies of *down-core concentrations* have been widely used when anthropogenic inputs to the environment should be described. In that case, down-core concentrations have been found to be very useful compared to other media as the sediment provides an integrated chemical picture over time and not a snapshot as, e.g., obtained from a single water sample. Thus, the down-core concentration technique is particularly useful when time trends over decades or centuries should be established. The technique can only be used on fine-grained sediments cored or sampled in areas with a continuous deposition. One should be aware that changes in sediment dynamics, changes in redox conditions, bioturbation as well as various diagenetic reactions may influence the concentration profile in the sediment. The dates at which particular sections of the core were deposited can be established using the concentration profiles of certain radioactive isotopes.

In addition to the factors above, it should be noted that the detection of a temporal trend from a contaminant concentration profile in a sediment core depends on the relative magnitudes of the sedimentation rate and the sediment mixing in the sediment surface layer (caused by bioturbation or other processes). That is, the minimum number of years needed to detect an increase or decrease of concentrations in the vertical profile (i.e., to detect a trend) is lower in a site with a high sedimentation rate together with limited surface mixing. The opposite situation, strong mixing together with a very low sedimentation rate in a site, would produce a concentration profile where detection of a trend may be difficult (too many years would be needed) or impossible (the case of a “straight”, “totally blurred” profile). Laminated sediments are usually an exception to this statement because they may occur in areas with no bioturbation (e.g., some areas in the Baltic Sea).

Although sediment cores, and dated sediments cores in particular, can provide a sensitive detection of time trends of contaminant concentrations at “one time”, as well as a great amount of other relevant information, suitable sampling sites are often difficult to find, particularly for coastal areas. The dating of cores is also a costly operation.

When sediment cores cannot be used and laminated sediments are not available, the *repeated surface sediment sampling* technique is recommended. The sampling strategy can be based on either repeated sampling of fixed stations or on random sampling methods. The time interval between the sampling and/or the thickness of the surface sample taken is dependent on both the sedimentation rate and the sediment mixing rate. The former can be measured and the latter calculated.

In highly dynamic areas dominated by sandy bottom sediments, it is not possible to take surface samples representing a given time span of deposition. Instead, surface samples from such bottom areas are to be considered as representing the current level of contamination recorded in the fine fraction of the sediments. No older information on contaminants is stored in these types of sediments. In these cases, time trend studies can be conducted by comparing average concentrations from repeated sampling of surface sediments. The sampling should be done within the same area but not necessarily on the same sites. By following either *a random sampling technique* or *a stratified random sampling technique,* the statistical probability can be ensured by different methods.

The *sediment trap* is a technique that has been used in trend monitoring in more limited areas. The traps collect material settling throughout the water column, but the influence of resuspension is strong for traps used in shallow areas with depths less than the maximum wave base depth. The advantage of the sediment trap method is the fixed time interval that can be used when collecting material. The technique may not give comparable results with bottom sediment sampling techniques. Another disadvantage of the technique is the cost of traps and that they may create problems and conflicts with other uses of the marine environment, e.g., trawling.

Questionnaire on sediment sampling

To assist the work of the OSPAR Working Group on Monitoring (MON), a questionnaire was distributed to the participating countries. The objective of this was to supply MON and ICES with an overview of monitoring activities on sediment currently carried out or planned to be implemented in the different countries. For each of the above-mentioned sediment sampling strategies, the number of stations, cm of sediment material considered, year of programme start or next sampling round, and whether data are reported to ICES are included, as well as the fraction analysed. The results of this questionnaire are shown in Table A2.1.

Analytical techniques

For metal analysis, the detection by ICP-MS, ICP-OES or AAS techniques are of similar quality, apart from results close to the varying detection limits of the different techniques. It is therefore mainly of interest whether the digestion/fusion which brought the sediment to a fluid phase for introduction to the detector was total or partial.

For the organics, especially PAHs, the extraction technique can also be crucial for the extraction efficiency, and the use of high temperature systems/continuous extraction methods compared to room temperature shaking methods can yield incomplete extraction for the latter, with the largest difference for the more hydrophobic substances.

Normalization

In order for the normalization guidelines to be used in transportation bottom areas, it is necessary to measure Al, Li, and TOC but also determine a pivotal point for each area, e.g., by analysing the sand grains after stripping of organic and clay particles. In sedimentation areas, the sediment usually can be regarded as "normalized" by nature, and a key normalization parameter in this case is the dry matter content.

Questionnaire on analytical techniques

To assist the work of the OSPAR Working Group on Monitoring (MON), a questionnaire was distributed to the participating countries. The objective of this was to supply MON and ICES with an overview of methods and parameters analysed within the monitoring programmes. The parameters and extraction techniques (as close as possible) are shown in Table A2.2.

Overview of data currently held in the ICES database

This overview is based on a total search in the ICES database as of March 2003, extracting all data where more than one data set is available. It should be noted that all available data, without regarding QA status or release dates, have been used, so this is an upper estimate of available data.

Table A2.3 gives an overview of the time span for each parameter group, i.e., metals, PAHs, PCBs and others, and when the last reporting of data to the data sets was done. However, the time span is not an indication of the number of years for which samples have been analysed, as some samples are taken only every five years. Also, as the search was based on laboratories, some of the time spans can be longer if stations were transferred between laboratories.

Table A2.1. Inventory of methods for sediment monitoring—Questionnaire on sampling strategy.

Country	Core samples/ retrospective monitoring	Surficial sediments, stations	Surficial sediments, regions	Suspended matter	Fraction used	Others (specify)
Belgium	No	22 st. 10 cm, yearly 1991/2003	No	No	<63 µm (all) <2 mm (metals)	
Denmark	No	49 st. 2 cm, 5 year, 2000/2003	No	No	<2 mm	No
France	8 st. ( ) 20–30 cm/1–3 cm, 10 years 1993/2003	Sediments move around	80−100 st.× 7 ( ) 1 cm, 10 years 1993–2003	No	<2 mm	One region sampled every year.
Finland	ca. 10 st ( ) 20 cm/1 cm 2003	No	No	Only research	<2 mm	
Germany	Not available at meeting	Not available at meeting	Not available at meeting	Not available at meeting	<20 µm (HM) (<63 µm organic)	Many institutes, different methods
Iceland	Not at meetings	Not at meetings	Not at meetings	Not at meetings	Not at meetings	Not at meetings
Ireland	No	Under devel. ( ) Stations in regions to be repeated	5 regions (× 15st. each) ( ) 1 cm, 5 years 2001–2003	No	<2 mm (HM historical data) <63 µm (organics and HM)	One region sampled every year.
Norway	Cores taken for some stations	15–20 st. 1–2 cm, 5/10 year 1986−1997/2003	Early programme 1980	No	<2 mm	Number of stations depending on funding, sampling interval dependent on sedimentation rates at each station
Portugal	ca. 5 st 20–50 cm, 5 cm 1985/project based	C: 32 st. yearly E: 120 st. 4 year 5 cm, 1980/2003	No	No	<2 mm all < 63 µm (HM) < 20 µm (HM, 2000)	Also project based C: Coastline, E: Estuaries

Under each heading: no indicates not applicable, number of stations (st.), if (to be) reported to ICES; for cores: target maximum depth and cm intervals for sampling; for surficial sediments: cm is centimetres of top layer sampled, and for both, years between sampling with first year/next year of sampling.

Table A2.1. Continued.

Country	Core samples/ retrospective monitoring	Surficial sediments, stations	Surficial sediments, regions	Suspended matter	Fraction used	Others (specify)
Spain	No	60 st. 5 cm, yearly 1994/2003	No	No	<63 µm HM <2 mm organics	
The Netherlands	Project based 30–50 cm/1 cm 2001	ca. 60 st. 5 cm, 3 year 1990/2003	No	8 st. ((( ))) SPM. 4 × year 1988	<63 µm	
United Kingdom	No	30-60 st. 1 cm, yearly 1999/	No	No	<63 µm all <2 mm if TT continues	Samples taken since 1994, but few analysed
Sweden	7 × 16 cores 80 cm/10 cm, 5 year 2003 upper 1 cm analysed or 5 upper laminas	16 stations. 1 cm, 5 year 1990/2003		One region, every 2 years 1989/2004	<2 mm	Also local coastal programmes. HELCOM-EGM programme. Cores stored for possible later analyses.
Poland		4 (7) stations 2003/2008			<2 mm	From HELCOM-EGM programme
Latvia		3 stations 2003/2008			<2 mm	From HELCOM-EGM programme

Under each heading: no indicates not applicable, number of stations (st.), if (to be) reported to ICES; for cores: target maximum depth and cm intervals for sampling; for surficial sediments: cm is centimetres of top layer sampled, and for both, years between sampling with first year/next year of sampling.

Table A2.2. Inventory of methods for national sediment monitoring—questionnaire on analytical strategy.

Country	Metals determined	Total/partial digestion	Organics determined	Extraction method	Normalizers and pivotal point (PP)
Belgium	Cd, Cu, Ni, Pb, Zn, Hg, As, Cr	Total (Partial Hg)	PAHs PCBs TBT (2003)	Soxhlet alk. sapon.	Al, Li, TOC, <63 µm PP: metals (2 size fr.)
Denmark	Cd, Cu, Ni, Pb, Zn, Hg	Total (Partial Hg, Cd)	PAHs, Nonylphenol, DEHP PCBs, DDTs, HCHs, TBT	Soxhlet Soxhlet Soxhlet ethylation+cold extraction	Al, Li, LOI, TOC, <63 µm PP: no
France	Pb, Cd, Cu, Zn, Hg, Co, Ni, Cr, Ti, V		PAHs PCBs, DDTs, HCHs		Al, Li, LOI, TOC, $CaCO_3$, <63 µm
Finland	Zn, Pb, Cd, Cu, Zn, As, Hg, Cr, Ni (possibly ICP-MS)	Total (Partial Hg)	PAHs PCBs, DDTs		Al, Li, TOC, <63µm PP: no plans
Germany	Different labs	Different labs	Different labs	Different labs	Different labs
Iceland					
Ireland	Cd, Cu, Ni, Pb, Zn, Hg, Cr	Total (Partial Hg)	PCBs, DDTs, HCHs, OCPs (Toxaphenes) PAH, TBT, Brominated flame retardants.	Soxhlet/solvent extraction Ext. lab.	Al, Li, LOI, TOC, <63 µm; PP: no
Norway	Zn, Pb, Cd, Cu, As, Hg, Cr, Ni	Total	PAHs PCBs, Possibly TBT	Soxhlet Soxhlet	Al, Li, TOC, <63 µm; PP: no
Portugal	Cd, Cu, Ni, Pb, Zn, Hg, Cr	Total (Partial Hg)	PAHs PCBs, THC	Soxhlet Soxhlet Direct extraction	Al, Li, Mn, Fe, TOC, LOI, <63 µm PP: metals (2 size fr.)
Spain	Cd, Cu, Ni, Pb, Zn, Hg	Partial (Aqua regia)	PAHs PCBs, DDTs, HCHs HCB	Soxhlet	Fe, Mn, TOC, <63 µm PP: no

Under each heading: no if not applicable; for metals, the individual metals (or number if more than ten); for organics, each group of organics and extraction methods; for normalization, each expected normalizer plus PP: status for pivotal points to be reported to ICES.

Table A2.2. Continued.

Country	Metals determined	Total/partial digestion	Organics determined	Extraction method	Normalizers and pivotal point (PP)
The Netherlands	Zn, Pb, Cd, Cu, Zn, As, Hg, Cr, Ni - now 70 elements	Partial	PAHs PCBs, TBT	Soxhlet (ASE) Soxhlet (ASE) ethylation+extraction	Al, Li, TOC, <63 and 2 µm PP: incl. uncertainty
United Kingdom	Zn, Pb, Cd, Cu, As, Hg, Cr, Ni	Total	PAHs PCBs		Al, Li, Fe, Mn, TOC, grain size PP: ??
Sweden	57 elements	Total (Partial Hg)	PAHs, PCBs, DDTs, HCHs, HCBs, chlordanes, TBT, Diphenylethers		Al, Li, TOC, grain size, $CaCO_3$ PP:??

Under each heading: no if not applicable; for metals, the individual metals (or number if more than ten); for organics, each group of organics and extraction methods; for normalization, each expected normalizer plus PP: status for pivotal points to be reported to ICES.

Table A2.3. Inventory of data held in the ICES Marine Data Centre.

Country	Years	Metals	PAHs	PCBs	Others
Belgium	**Maximum 10**	**13**	**2**	**18**	**14**
2000	From 1–4 years	4	2	10	4
	From 5–6 years	1	0	5	2
	More than 6 years	8	0	5	8
Denmark	**Maximum 6**	**2**	**0**	**0**	**2**
1991	From 1–4 years	1	0	0	1
	More than 6 years	1	0	0	1
Germany	**Maximum 16**	**77**	**10**	**22**	**79**
2001	From 1–4 years	20	4	4	22
	From 5–6 years	11	6	9	11
	More than 6 years	46	0	9	46
Ireland	**Maximum 3**	**4**	**0**	**2**	**4**
1995	From 1–4 years	4	0	2	4
Netherlands	**Maximum 6**	**7**	**7**	**7**	**7**
1990	From 1–4 years	1	1	1	1
	From 5–6 years	6	6	6	6
Norway	**Maximum 11**	**11**	**11**	**6**	**14**
2000	From 1–4 years	1	4	1	4
	From 5–6 years	3	3	0	3
	More than 6 years	7	4	5	7
Poland	**Maximum 1**	**0**	**0**	**1**	**1**
1990	From 1–4 years	0	0	1	1
Portugal	**Maximum 6**	**6**	**0**	**0**	**6**
1990	From 5–6 years	6	0	0	6
United Kingdom	**Maximum 7**	**43**	**0**	**28**	**25**
1999	From 1–4 years	23	0	11	25
	From 5–6 years	7	0	6	0
	More than 6 years	13	0	11	0

The first **(bold)** line indicates the maximum number of years and the total number of samples for each country, with the last year of reporting indicated in the second line.

The next lines contain the number of stations with data for 1–4 years, 5–6 years, and 7 and more years. The years are defined as the span between the first and last years sampled, e.g., two samplings with five-year intervals will end up as one data set in the "From 5–6 years" row.

ANNEX 3: INVENTORY OF SEDIMENT QUALITY CRITERIA IN ICES MEMBER COUNTRIES

I INTRODUCTION

At the meeting of the Working Group on Marine Sediments in Relation to Pollution (WGMS) in 2001, it was recommended that WGMS produce an inventory of national sediment quality criteria and the approaches used to reach such criteria.

Objective

Sediment quality guidelines (SQG) provide a method for assessing the quality of sediments, and may be used in classifying environmental quality or determining whether dredged material may be disposed of at sea. This inventory identifies the situation in ICES Member Countries in 2002. ICES can provide scientific advice on the approaches used, so that managers may take more informed decisions based on the use of such criteria.

Summary

A review of approaches used towards sediment quality guidelines (sometimes referred to as sediment quality criteria) was performed. Approaches for the application of environmental quality standards and dredged material disposal were also considered. The general approaches taken by member countries are summarized in Tables A3.1 and A3.4. Where data are available for contaminant limits, these have been included in this report.

Most member countries currently do not have environmental quality standards (EQS) for sediments set in legislation. In these cases, responsible authorities frequently use guideline values, which may be based on the OSPAR background/reference concentration for that substance, or on locally derived background concentrations. However, several member countries are developing EQS values for sediments and more data should become available over the next few years.

In contrast, most countries do have legislative standards governing the disposal of dredged material at sea. Most member countries operate an action level approach to the disposal of dredged material, where "target" values are used to represent (near-) background concentrations and "limit" (or "intervention") values represent the upper limit of acceptability above which action may have to be taken. Three countries use a case-by-case approach to dredged material assessment, but these are under review and are likely to be replaced by some type of action level approach.

For most substances considered, there is some consistency among the concentrations set for target concentration values. Ranges in limit values are much larger, however, and this cannot generally be explained by differences in grain size. It seems more likely that the large differences between limit values in different countries instead reflect local conditions.

II REVIEW OF SOME METHODS USED FOR SETTING SEDIMENT QUALITY GUIDELINES

Several different approaches have been used for setting sediment quality guidelines (SQGs). These are listed in this section together with the advantages and disadvantages of each method. Ingersoll *et al.* (1997) gave helpful criteria for evaluating SQGs. As a minimum, reliable and useful SQGs should provide tools that are predictive of the presence of toxicity or other effects, and which demonstrate a dose-response relationship to chemical concentrations calibrated to the SQGs.

1 Background sediment chemistry

The background sediment chemistry approach is based on a comparison of concentrations of metals in contaminated sediments with those in reference, uncontaminated sediments.

Advantages:

- can be implemented using available data;
- has minimal data requirements because of its simplicity;
- does not require toxicity testing or detailed chemical reasoning.

Disadvantages:

- difficult to define reference sediments;
- highly site-specific;
- does not take bioavailability into account;
- difficult to defend because of the difficulties of identifying background sediments.

2 Water Quality Criteria (WQC)

The water quality criteria approach measures the concentrations of metals in the interstitial waters of sediments and compares the values with accepted water quality standards.

Advantages:

- makes use of well-established toxicological database.

Disadvantages:

- assumes that exposure of sediment organisms is via interstitial water and that this is the main uptake route (i.e., does not take account of uptake of contaminant particles or uptake via food);

- difficult to measure interstitial water quality;
- does not relate to mixtures;
- does not relate to the sediment of interest;
- are of no use if there are no relevant WQCs;
- assumes that sediment infauna have the same sensitivity as other living organisms.

3 Sediment/water equilibrium partitioning

This approach combines Environmental Protection Agency (EPA) (or other) water quality criteria together with equilibrium partitioning calculations to obtain sediment contaminant concentrations that give rise to water concentrations equivalent to the criterion.

Advantages:

- makes use of well-established toxicological database;
- makes use of organic carbon;
- makes use of chemical equilibria which are often well-known;
- efficient for determining which chemicals are likely contributors to toxicity.

Disadvantages:

- assumes that interstitial water is the main uptake route (i.e., does not take account of uptake of contaminant particles or uptake via food);
- some partition coefficients are uncertain;
- only strictly valid for some organic compounds;
- data do not relate to mixtures;
- assumes that sediment infauna have the same sensitivity as other living organisms;
- does not account for the presence of mixtures;
- produces a single SQG number, thus failing to recognize the fact that data from real sediments are too uncertain to support more than a range of values.

4 Sediment bioassay

Sediment bioassays may be used in two ways. Firstly, test animals may be exposed to a range of sediments from clean to heavily contaminated areas. The response of the organism considered to be unacceptable is related to the sediment composition, which is then taken as the standard. Secondly, animals may be exposed to a range of spiked sediments. Again, the response of the organism considered to be unacceptable may be related to the sediment composition, which is then taken as the standard. In this second case, a dose-response curve is developed.

Advantages:

- similar to WQCs: technically acceptable and legally defensible;
- good for identifying problem sediments;
- deals with synergism in real-world sediments;
- does not require prior knowledge of mechanisms of uptake.

Disadvantages:

- uncertainty over the cause of any effect;
- difficult to implement with a range of organisms;
- difficult to dose sediments;
- may not reflect chronic effects, and chronic effects difficult to predict from acute endpoints;
- massive amount of work to examine mixtures and dosages which can occur;
- no basis at present for extrapolating to no-effect concentrations in sedimentary communities.

5 Effects range- and effects level- approach (i.e., ERL/ERM and PEL/TEL)

This approach uses statistical analyses of matching chemical and biological data which have been used to determine the concentration below which effects are rarely observed, and the concentration above which the incidence of effects is elevated.

Advantages:

- can be used with any chemical constituent;
- can use existing databases;
- does not require prior knowledge of mechanisms.

Disadvantages:

- large amount of field data required;
- values are potentially sensitive to data used in calculation;
- cannot separate contaminant effects.

6 Apparent effects threshold (AET)

This approach uses field data on chemical concentrations in sediments and at least one indicator of bioavailability/bioeffects (e.g., sediment bioassays, benthic infaunal community structure, bottom-fish histopathological abnormalities, bioaccumulation). It determines the concentration of a particular contaminant above which statistically significant biological effects (relative to a reference site) are always expected.

Advantages:

- can be used to develop criteria for any contaminant using any effects measure provided that it can be statistically evaluated;
- does not require prior knowledge of mechanisms;
- biological effects always occur above AET, so it is not uncertain.

Disadvantages:

- requires large database;
- can be influenced strongly by unknown toxic compounds;
- bioassay methods may not reflect chronic effects;

- no mechanism has been established to separate individual effects;
- not conservative (some toxic sediments will be missed);
- does not establish safe levels;
- produces a single SQG number, thus failing to recognize the fact that data from real sediments are too uncertain to support more than a range of values.

7 Sediment Quality Triad

This approach is based on correspondences between three measures: sediment chemistry to determine contamination, sediment bioassays to determine toxicity, and *in situ* bioeffects to determine alteration of resident communities.

Advantages:

- uses a combination of three different measures;
- does not require prior assumptions about mechanisms;
- can be used for any contaminant;
- accounts for both acute and chronic effects.

Disadvantages:

- requires a large database;
- can be strongly influenced by unmeasured compounds;
- statistical criteria have not yet been developed;
- methodology not yet fully developed.

8 Ecotoxicological approaches to sediment quality criteria

Perhaps the most developed of the ecotoxicological approaches to sediment quality criteria have been produced by Long and MacDonald (Long *et al.*, 1995; Long and MacDonald, 1998) in the U.S. Their data sets, which took a statistical approach in matching biological and chemical data from modelling, laboratory and field studies in North America, have since been developed to set sediment quality guidelines in a number of countries, notably the U.S., Canada, and Hong Kong. Chemical concentrations only were used to derive effects range-low (ERL) and effects range-median (ERM) sediment quality criteria. However, calculation of Threshold Effects Levels (TEL) and Probable Effects Levels (PEL) values incorporated concentrations associated with both effects and no observed effects (Long and MacDonald, 1998).

Some work has been reported assessing the "accuracy" of sediment quality guidelines. ERL and ERM values were used to define concentration ranges that were: 1) rarely (e.g., <10%); 2) occasionally (usually ~25%); or 3) frequently (usually ~70%) associated with adverse effects (Long and MacDonald, 1998). To assist in the management of prioritization of sediments, these authors reported results from studies where multiple exceedances of ERMs were observed. Their data showed, for example, that 40–60% of samples in which individual ERMs or PELs were exceeded proved to be highly toxic in amphipod survival tests. With respect to mixtures, where >10 ERMs (or >21 PELs) were exceeded, the probability of observing toxicity in amphipod survival tests was 74–88%. Where concentrations of substances were below the ERL or TEL, the incidence of effects was generally below 16% (Long and MacDonald, 1998). Such data provide confidence in the effectiveness of the ERM and PEL approach.

In a recent paper, Burton (2002) emphasized that sediment quality guidelines should be used in a screening manner as part of a holistic assessment. This integrated approach is in accordance with the intended application of the action level guidelines currently being proposed. Balancing multiple "lines of evidence" concerning ecological assessment to aid decision-making is the focus of recent discussions regarding "weight of evidence" approaches to environmental management of sediments (Burton, 2002; Chapman *et al.*, 2002).

9 Discussion

It is apparent from the above that no single method for setting SQGs is free of problems. Several (e.g., background sediment chemistry, water quality criteria, equilibrium partitioning) are too simplistic because they either make no allowance for variations in bioavailability caused by differing sediment conditions, or assume incorrectly that all exposure occurs via the interstitial water. There is no doubt that the ingestion of sedimentary particles is a significant route by which adsorbed contaminants exert toxicity, so methods which ignore this are only applicable to non-benthic organisms.

In theory, a much better approach would be to use sediment bioassays as the basis for setting SQGs, because they measure sediment toxicity to real sedimentary organisms and the contaminants are present in a reasonably natural manner. However, it has to be recognized that spiked sediment tests do not mimic perfectly the bioavailability of some contaminants which have entered sediments naturally. Furthermore, there are only a few sediment toxicity test procedures available, with a relatively small number of taxa, and very few chronic test methods. This would severely limit the reliability of SQG-setting based on safety factors because an insufficient number of species and endpoints are currently available, potentially introducing the use of unrealistically high safety factors.

The Sediment Quality Triad is a more promising approach as it is based on the correspondence of chemical, bioassay, and *in situ* biological data from a given site. However, the methodology for integrating the data to give a reliable weight-of-evidence is not fully developed and the approach is probably too data-hungry and cumbersome to form the basis for operational sediment assessment for more than a handful of substances and locations. The remaining methods are all variations of the co-occurrence concept. These approaches make use of all available data from many sediment studies, and they have the advantage that the detailed methodology has been fully worked out and to some extent validated. The Apparent Effects Threshold is the least attractive of the co-occurrence procedures because it is insufficiently precautionary. The other two main methods (TEL/PEL and ERL/ERM) are very similar and sufficiently conservative, with little to choose between them. However, the ERL/ERM is simpler to operate and has received the most validation. Of the available methods, it is therefore considered to be the most promising.

10 References

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III EXISTING APPROACHES IN MEMBER COUNTRIES

IIIa Environmental Quality Standards for Sediments

Currently, countries frequently compare their sediment data with those specified under OSPAR background/reference concentrations (BRC) (MON 00/5/Info.4-E). However, implementation of the Water Framework Directive (WFD) (Directive 2000/60/EC) is causing some member countries to review their existing approach, since the WFD requires definition of five class levels (high status, good, moderate, poor, and bad). An EU group, the Expert Advisory Forum (EAF), is examining Environmental Quality Standards (EQS) for priority substances, in line with the requirements for the WFD, and these EQS will define the boundaries between good and moderate status.

The approach for the other classes has yet to be considered. However, the WFD defines high status for specific synthetic substances as close to zero and at least below the limit of detection of the most advanced techniques in general use. For specific non-synthetic substances, concentrations should remain within the range normally associated with undisturbed conditions.

A summary of criteria legislation status, and of the approaches towards environmental quality standards and guidelines for sediments used in member countries is given in Table A3.1. See Tables A3.6–A3.14 for more detailed data applied in Canada, The Netherlands, Norway, and Sweden.

Most member countries currently do not have environmental quality standards for sediments set in legislation. In these cases, responsible authorities frequently use guideline values, which may be based on the OSPAR background/reference concentration for that substance, or on locally derived background concentrations. However, several member countries are developing EQS values for sediments and more data should become available over the next few years.

Table A3.2 shows the range in EQS values currently available. The range for the five-class system used by both Norway and Sweden shows the lowest value given in that class to the highest value, providing an indication as to the range of concentrations that may be graded as, for example, “class 4”.

It can be difficult to compare between differing classification schemes, and this difficulty may be compounded as more countries adopt EQS values, depending upon the approach taken to determining environmental quality, protection of aquatic life, etc. However, broadly it can be seen that the Netherlands No Effects Concentration (NEC) and Canadian ISQG correspond to Classes 1–2 under the Norwegian and Swedish classifications. However, the ISQG values for benzo[*a*]pyrene and DDTs are somewhat higher, being closer to the Dutch MPC and classes 4–5 (the disparity in DDTs is unlikely to result from the inclusion of *o,p'* isomers in the Canadian values). The Maximum Permissible Concentration (MPC) and PEL concentrations generally fall in Swedish class 4–5, but between classes 2–5 in the Norwegian system, which appears to result from the large range covered by the Norwegian values.

Table A3.1. Summary of criteria legislation status, and approaches towards Environmental Quality Standards and Guidelines for sediments in ICES Member Countries.

Country	Legislation	Notes
Belgium	x	Data compared against OSPAR BRC.
Canada	?	ISQG and PEL approach, from field data demonstrating associations between chemistry and biological effects.
Denmark	x	Locally derived BCs from reference sediments (mainly for metals).
Finland	x	
France	x	Data compared against OSPAR BRC and locally derived BC from cored sediments.
Germany	x	
Portugal	x	Data compared against OSPAR BRC.
Netherlands	yes	Two-class system (NEC and MPC) based on equilibrium partitioning or ecotoxicological data. See further notes below.
Norway	yes	Five-class system. Data compared against local, surface sediment concentrations. See further notes below.
Ireland	x	
Spain	x	Data compared with international (e.g., USEPA) and locally derived BCs.
Sweden	yes	Five-class system. BCs derived from cored sediments. See further notes below.
UK (England and Wales)	x	Data compared against OSPAR BRC; locally derived BCs in preparation.

Notes:

x = not currently covered by legislation;
ISQG = Interim sediment quality guideline;
USEPA = United States Environmental Protection Agency;
NEC = No Effect Concentration;
BRC = Background/reference concentration;
PEL = Probable effects level;
BC = background concentration;
MPC = Maximum Permissible Concentration.

Further notes to Table A3.1 for individual countries

CANADA

Further details on Canada's approach to sediment quality guidelines can be found at http://www.ec.gc.ca/ceqg-rcqe/English/Ceqg/Sediment/default.cfm and data are presented in Table A3.6.

Sediment quality guidelines formulated on the basis of biological effects data on sediment-associated chemicals are intended to be used as nationally consistent benchmarks. During their implementation, however, allowance must be made for the incidence of natural inorganic and organic substances in sediments. Adverse biological effects may be observed below measured chemical concentrations that are attributable to natural enrichment. However, management concerns over the potential for adverse effects of sediment-associated chemicals (particularly trace metals) must be practically focused on those chemicals whose concentrations have been augmented above those that would be expected to occur naturally. Therefore, the potential for adverse biological effects as indicated by the exceedances of SQGs must be evaluated in conjunction with other information such as the natural background concentrations of substances. In some management scenarios, it may also be necessary to consider concentrations of ubiquitous organic chemicals (i.e., the low level contamination of certain substances that are found throughout many environmental compartments) that are representative of reference or "clean" sites.

Guiding Principles

The following guiding principles for the development of Canadian SQGs for the protection of aquatic life are based on those adopted by the Canadian Council of Ministers of the Environment (CCME, 1991) for the development of Canadian water quality guidelines.

SQGs are numerical concentrations or narrative statements that are set with the intention of protecting all forms of aquatic life and all aspects of their aquatic life cycles during an indefinite period of exposure to substances associated with bed sediments.

Table A3.2. Summary of value ranges used for classifying sediments according to EQS values.

Chemical	Units	Norway and Sweden					Netherlands		Canada	
		Class 1	Class 2	Class 3	Class 4	Class 5	NEC	MPC	ISQG	PEL
As	mg kg^{-1}	≤10–<20	10–80	16–400	26–1,000	>40–>1,000	29	55	7.24	41.6
Cd	mg kg^{-1}	≤0.2–<0.25	0.2–1	0.5–5	1.2–10	>3–>10	0.8	12	0.7	4.2
Cr	mg kg^{-1}	<70–≤80	70–300	112–1,500	160–5,000	>224–>5,000	100	380	52.3	160
Cu	mg kg^{-1}	≤15–<35	15–150	30–700	60–1,500	>120–>1,500	36	73	18.7	108
Inorganic Hg	mg kg^{-1}	*	*	*	*	*	0.3	10	*	*
Organic Hg	mg kg^{-1}	*	*	*	*	*	0.3	1.4	*	*
Hg	mg kg^{-1}	≤0.04–<0.15	0.04–0.6	0.1–3	0.27–5	>0.72–>5	*	*	0.13	0.7
Pb	mg kg^{-1}	<30–≤31	30–120	47–600	68–1,500	>102–>1,500	85	530	30.2	112
Ni	mg kg^{-1}	<30–≤33	30–130	43–600	56–1,500	>79–>1,500	35	44	*	*
Zn	mg kg^{-1}	≤85–<150	85–650	128–3,000	196–10,000	>298–>10,000	140	520	124	271
Σ7 PCBs	µg kg^{-1}	0–<5	0–25	1.3–100	4–300	>15–>300	*	*	(21.5)[3]	(189)[3]
B[*a*]P	µg kg^{-1}	0–<10	0–50	20–200	60–500	>180–>500	3	3000	88.8	763
EPOCl[1]	µg kg^{-1}	0–<100	0–500	150–2,000	700–15,000	>3,000–>15,000	*	*	*	*
HCB	µg kg^{-1}	0–<0.5	0–2.5	0.04–10	0.2–50	>1–>50	0.05	5	*	*
Sum DDT	µg kg^{-1}	0–<0.5	0–2.5	0.2–10	1–50	>6–>50	*	*	*	*
p,p′-DDT[2]	µg kg^{-1}	0	0–0.02	0.02–0.1	0.1–0.7	> 0.7	0.09	9	1.19[4]	4.77[4]
p,p′-DDE[2]	µg kg^{-1}	0	0–0.2	0.2–0.7	0.7–2.5	> 2.5	0.02	2	2.07[4]	374[4]
p,p′-DDD[2]	µg kg^{-1}	0	0–0.13	0.13–0.8	0.8–5	> 5	0.01	1	1.22[4]	7.81[4]

Notes
[1] EPOCl = Total persistent extractable organic chlorine.
[2] Values for Swedish five classes only.
[3] Total PCBs.
[4] Total of *p,p'* and *o,p'* isomers.

In deriving SQGs for the protection of aquatic life, all components of the aquatic ecosystem (e.g., bacteria, algae, macrophytes, invertebrates, fish) are considered, if the data are available. However, evaluation of the available data should focus on ecologically relevant species.

Interim SQGs (ISQGs) are derived when data are available but limited, and information gaps are explicitly outlined.

Unless otherwise specified, SQGs refer to the total concentration of the substance in surficial sediments (i.e., the upper few centimetres) on a dry weight basis (e.g., mg kg^{-1} dry weight). However, sediments represent a complex and dynamic matrix of biotic and abiotic components that may influence the bioavailability of sediment-associated chemicals. When sufficient information is available to define the influence of any factor on the toxicity of a specific substance (e.g., TOC for non-polar organic substances) (Swartz *et al*., 1990; Di Toro *et al*., 1991), the guidelines will be developed to reflect this relationship. Consideration of these relationships will increase the applicability of guidelines to a wide variety of sediments throughout Canada.

SQGs are refined as new and relevant scientific data become available. The refinement of these guidelines in the longer term will provide a means of ensuring their broader applicability.

References

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Table A3.3. Comparison of terms used in the setting of ERLs and EQSs in the Netherlands.

Description	ERL	EQS
The NC (negligible concentration) represents a value causing negligible effects to ecosystems. The NC is derived from the MPC by dividing it by 100. This factor is applied to take into account possible synergistic effects.	NC, Negligible concentration. (air, water, soil, groundwater, and sediment)	Target value (air, water, soil, groundwater, and sediment)
A concentration of a substance in air, water, soil or sediment that should protect all species in ecosystems from adverse effects of that substance. A cut-off value is set at the fifth percentile if a species sensitivity distribution of NOECs is used. This is the hazardous concentration of 5% of the species (HC5).	MPC, maximum permissible concentration. (air, water, soil, groundwater, and sediment)	MPC, maximum permissible concentration. (air, water, groundwater, and sediment)
A concentration of a substance in the soil, sediment, and groundwater at which functions in these compartments will be seriously affected or are threatened to be negatively affected. This is assumed to occur when 50% of the species and/or 50% of the microbial and enzymatic processes are possibly affected.	SRC eco, Serious Risk Concentration for the ecosystem. (water, soil, groundwater, and sediment)	Intervention value* based on SRC eco (water, soil, groundwater, and sediment) if this value is lower than SRC_{human}. SRC_{human} is derived elsewhere.

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DENMARK

Sediment quality criteria have been requested under the Water Framework Directive, and it is expected that this will guide the development of such EQS in Denmark within the next few years.

FINLAND

Sediment quality criteria have been requested under the Water Framework Directive, and it is expected that this will guide the development of such environmental quality criteria in Finland.

THE NETHERLANDS

Table A3.7 shows NC (Negligible Concentration) and MPC (Maximum Permissible Concentration) environmental quality standard data for sediments in the Netherlands.

Derivations of ERLs and EQSs in the Netherlands (Traas, 2001)

Introduction

ERLs (environmental risk limits) are derived for different environmental compartments, based on observed or expected effects on species inhabiting these compartments, including effects from food chain exposure of predators (secondary poisoning). ERLs are used as scientific advisory values to set EQSs (environmental quality standards) by the government. When setting EQSs, the government can take into consideration the advice of consulting parties and can take into account socio-economic factors. Table A3.3 explains the relationship between the different ERLs and EQSs: the EQS target value can be related to Action Level 1 in the classification system of some other countries.

Derivation of ERLs

ERLs are derived using single-species toxicity data or processes for soil and physico-chemical characteristics, with different approaches depending on the amount of information available. When chronic toxicity data (NOEC = no observed effect concentration) for four or more species of at least four different taxonomic groups are available for a particular environmental compartment, a statistical procedure is applied to derive ERLs (Aldenberg and Jaworska, 2000). This approach is called refined effect assessment. The basic assumption of the method is that the log of the sensitivities of a set of species in a community can usually be described by a Normal distribution. The available ecotoxicological data are seen as a sample from this distribution and are used to estimate the parameters of the species sensitivity distribution. Specific percentiles of this species sensitivity distribution are chosen to determine ERLs (5th percentile = MPC, 50th percentile = SRCeco).

When fewer data are available a set of assessment factors is applied, varying from 10 to 1000, depending on the type of data. This approach is called the preliminary effect assessment. Assessment factors from the technical guidance document (TGD) of the European Union are used (ECB, 1996). Currently the TGD is being updated and a separate chapter on the effect assessment of substances in marine water will be added. If the base set for using the TGD method is incomplete, the modified EPA method is used.

Equilibrium partitioning

If toxicity data for species representative of soil and sediment cannot be found or are insufficient, equilibrium partitioning is applied to derive ERLs for soil and sediment from the ERL for water. In this case, soil/water or sediment/water partition coefficients are required. ERL soil and sediment are calculated according to:

1. ERL (sed/soil) = ERL (water) × Kp
2. ERL (sed/soil) = environmental risk limit for terrestrial species using EqP method in mg kg^{-1}
3. ERL (water) = environmental risk limit for aquatic species in mg l^{-1}
4. Kp = partition coefficient for standard soil or standard sediment.

Secondary poisoning

Substances that accumulate through the food chain may exert toxic effects on higher organisms such as birds and mammals. If a substance is potentially bioaccumulative, toxicological data for birds and mammals are collected. Dividing the data by bioaccumulation data (BCF) and using several conversion factors leads to a NOEC water for biota. These NOECs are combined with the available direct toxicity data and the ERLs are calculated according to the methods described above (for details, see Traas, 2001).

<u>The added risk approach</u>

For naturally occurring substances, background concentrations (BC) are taken into account. Maximum Permissible Addition (MPA) are calculated using a similar approach as the MPC for substances having no natural background concentration. The MPC is calculated according to the equation:

MPC = MPA + BC

The negligible concentration is calculated according to:

NC = NA + BC, where NA = MPA/100

Standardization

ERLs for soil and sediment are calculated for a standardized soil, i.e., for metals, a soil or sediment that contains 10% organic matter and 25% clay; for organic substances, one that contains 10% organic matter. ERLs for water are reported for dissolved and total concentrations (including a standard amount of suspended matter) and distinguished between freshwater and saltwater if values are significantly different.

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NORWAY

The Norwegian sediment criteria for Classification of Environmental Quality and Degree of Pollution (CEQDP) in fjords and coastal waters represent the basis for assessing environmental quality (see Table A3.8 for values). The EQS are divided into five classifications, with the lowest concentration level being based on nationally derived background values. These background values were developed from surface samples from areas which had no point source contamination, but which were not necessarily free of anthropogenic influence, so that they are not intended to represent pre-industrial values, but rather represent concentrations seen today in uncontaminated areas. There are plans to review the classification.

SPAIN

One method for deriving the degree of metal contamination in sediments, proposed by Tomlinson *et al.* (1980), follows the approach below. The levels of contamination for metals are estimated from the natural background of metals in the Basque coastline (AZTI, unpublished data) and follow the Müller criteria for sediment classification (Müller, 1979).

LPI = (EF1× EF2 × EF3.....EFn)1/n
EF = Cn/Bn
LPI: Load Pollution Index
EF: Enrichment Factor
Cn: metal concentration
Bn: metal background concentration

LPI > 48: extreme contamination (EC)

48 > LPI > 12: strong contamination (SC)

12 > LPI > 3: moderate contamination (C)

3 > LPI > 1: slight contamination (LC)

LPI < 1: no contamination (NC)

The levels of contamination for organic compounds are estimated on the basis of:

• If there are reference values (PCBs, DDTs, PAHs)

NC < detection limit < **LC** < guide or low toxicity < **C** < limit or median toxicity < **SC**

Where

NC = non-contaminated,
LC = low contamination,
C = contaminated, and SC = serious contamination.

• If there are no reference values (HCB, aldrin, dieldrin, t-nonaclor, HCH)

NC < detection limit < **LC** < 3 × detection limit < **C** < 12 × detection limit < **SC**

In Huelva, the sediment quality triad approach is also used in surveillance following the Aznalcollar disaster of 1997.

References

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Tomlinson, D.L, Wilson, J.G., Marris, C.R., and Jeffrey, D.W. 1980. Problems in the assessment of heavy metal levels in estuaries and the formation of pollution index. Helgolander Meeresuntersuchungen, 33: 566–575.

SWEDEN

Introduction

In 1999 the Swedish Environmental Protection Agency established the Swedish Environmental Quality Criteria (EQC) for marine sediments, among others (Anon., 1999). The purpose of these criteria is to enable local and regional authorities and others to make accurate assessments of environmental quality on the basis of available data on the state of the environment and thus obtain a better basis for environmental planning and management by objectives. The assessment involves two aspects: one dealing with the appraisal of whether the recorded state has any adverse effects on the environment or human health, the other with appraisal of the extent to which the recorded state deviates from a "background value". In most cases, the background value represents an estimate of a "natural" state. The results of both appraisals are expressed on a scale of 1–5 (five classes). The criteria on coasts and seas are a tool for determining the environmental quality of the marine environment in terms of three threats to the environment:

- eutrophication;
- toxic organic micro-pollutants and metals;
- physical disturbance (exploitation of the coast).

It is difficult to make a definitive assessment of changes in marine biodiversity at present. The Swedish environmental quality is therefore assessed on the basis of the environmental threats affecting the prospects of preserving biodiversity. The parameters used to assess eutrophication are, among others, nutrients, macrofauna, and flora. Metals and toxic organic micro-pollutants are assessed in sediments and organisms. The following text deals with the Swedish criteria in sediments.

Organic micro-pollutants

The Swedish classification is not based on the effects on biota, since at present there is a large gap in knowledge of the effects. Instead the classification of organic micro-pollutants in sediments is based on the statistical distribution of a large data set on the superficial sediments (0–1 cm). This data set comes from the Geological Survey of Sweden, which during the past fifteen years has systematically collected sediment samples from the Swedish coastal and sea areas and analysed them using the same methods throughout. The classification system for organic contaminants is shown in Table A3.9. These classes have been derived from concentrations found in Swedish waters. Class 1 is set at zero since the organic contaminants do not exist naturally. Further notes on the determination of class boundaries are given with the table.

Metals

The assessment of metals is based on the deviation from a background value representing pre-industrial concentration levels. The background value (Table A3.10) comes from a Normally distributed data set of reference samples collected by the Geological Survey of Sweden in Swedish coastal and sea areas. The reference samples were taken at about 55 cm depth of burial, i.e., where the concentrations are believed to represent those from the pre-industrial period. From the data set, the 50th percentiles have been used as the background values. Natural concentrations of metals in sediments are rather similar along the Swedish coast, although local discrepancies may exist, as for example in the Bothnian Bay off the ore fields. In such cases, a local pre-industrial value may be used as the background value.

Tables A3.11 and A3.12 show the deviation from background values. Table A3.11 shows background values derived from partial digestion (7 M HNO_3) of the sediment following the Swedish standard used in the country since the beginning of the 1970s. Table A3.12 shows background values based on the digestion of the total sediment (HF or $LiBO_4$-fusion).

One objective of the classification system is that the system should make it possible to detect whether a local point source exists and is affecting a restricted sea area. Thus, the boundary between the classes 4 and 5 is put at a level where point sources clearly influence the concentration. Tables A3.13 and A3.14 present the concentration data for the Swedish classification system of sediments. The boundary between classes 1 and 2 is set at a ratio of 1, i.e., when the environmental state is equal to the background value, so that the class boundary between classes 1 and 2 should represent a "natural" concentration. The calculated boundaries are based on the background values given in Table A3.10 and the factors given in Tables A3.11 and A3.12. The classes 2, 3, and 4 are intended to successively show the effect of increased diffuse pollution. Table A3.13, using the Swedish standard method, sets the boundary between the classes 4 and 5 at the 95th percentile of a data set from locally unaffected coastal areas (for nickel and chromium the 99th percentile is used). Table A3.14 for total analyses sets the boundary at the 99th percentile of offshore data. Further notes on the determination of class boundaries are given with the tables.

Reference

Anon. 1999. Bedömningsgrunder för miljökvalitet – Kust och hav (Assessments of Environmental Quality – in Coast and Sea). Swedish Environmental Protection Agency, Report No. 4914. 134 pp.

UNITED KINGDOM

The UK does not currently have any standards for the assessment of the general environmental quality of sediments. Data collected under the National Marine Monitoring Programme are currently assessed against the available Background/Reference Concentrations as recommended by OSPAR. A review is currently under way to establish a more relevant set of BRC data for the UK.

IIIb Dredged Material Standards

Dredged material disposal is governed by legislation in most countries. The most common approach is a two action level approach, where three concentration classes are defined. Contaminants are measured in a particular size fraction and concentrations compared against limit values in each category:

Category 1: $C < AL\ 1$
Category 2: $AL\ 1 < C < AL\ 2$
Category 3: $C > AL\ 2$

C = concentration
AL = action level

Concentrations in category 1 (or "target level") mean that disposal would generally be permitted (subject to consideration of other factors, e.g., volumes, grain size, etc.); those in category 2 indicate moderate contamination and the material would require further study before disposal could be permitted; those in category 3 ("limit" or "intervention" level) represent concentrations so high that disposal at sea would generally not be permitted. Table A3.4 summarizes the approaches used in dredged material assessment.

Portugal uses a five-category system for dredged material assessment, developed from environmental quality standards (Table A3.17). The Netherlands has one overall limit level, above which material may not be disposed of at sea, but also has various sub-levels which are applicable in particular situations (Table A3.18). A few countries operate on a "case-by-case" system, where each application for dredged material disposal is considered individually and internal guidelines may be applied by the licensing authority (i.e., standards which have no basis in law). In most such instances, the procedure is under review and is likely to be replaced by an action level approach.

Sediment chemistry is the main method by which assessment of the dredged material is made, although some countries are introducing ecotoxicological measures (e.g., Belgium, Germany).

Comparison of standards for assessment of dredged material disposal at sea

Table A3.5 summarizes the range of contaminant concentrations used in the assessment of dredged material, while Tables A3.15–A3.19 report the details of these values.

For most substances considered, there is some consistency among the concentrations set for target concentration values. Ranges in limit values are much larger, however, and this cannot generally be explained by differences in grain size. It seems more likely that the large differences between limit values in different countries instead reflect local conditions.

Table A3.4. Summary of approaches used for dredged material assessment.

Country	General approaches used	No. of categories in action level approach	Methods used in development of action levels	Notes
Belgium	Action level	3	1) sediment chemistry 2) bioassays	Quality criteria based on mean contaminant concentration in marine navigation channels
Denmark	Action level (see notes)	3	sediment chemistry	Not yet implemented in law
Finland	Case-by-case	---	---	Legislation for sediment quality criteria in preparation
France	Action level	3	sediment chemistry	
Germany	Action level	3	1) sediment chemistry 2) bioassays	
Portugal	Action level and case-by-case	5	sediment chemistry	Physical, biochemical, biological, toxicological, and persistence properties of the dredged material are analysed in a case-by-case approach
Netherlands	Action level	1 limit level*	sediment chemistry	Biological effects methods being tested. *Sub-levels for particular situations depending upon use
Norway	Action level + case-by-case	5	sediment chemistry	Categorization is the same as used for environmental quality standards
Ireland	Case-by-case: Action levels under development	---	---	Three-category action level approach under development
Spain	Action level	3	sediment chemistry	Sediment chemistry not yet implemented in law. Sediment bioassays under development
Sweden	?			
UK	England and Wales (E+W): Case-by-case approach: Action levels to be implemented shortly	---		E+W: Three-category action level approach in preparation. Scotland: Data assessed against OSPAR BRCs

Table A3.5. Summary of concentration ranges used in the assessment of dredged material for disposal at sea.

		TARGET VALUES			LIMIT VALUES		
Contaminant	Units	Range	Class range[1] (Portugal)	Maximum value in <2 mm fraction[2]	Range	Class range[1] (Portugal)	Maximum value in <2 mm fraction[2]
As	mg kg^{-1}	20–80	2	*	50–1,000	3 –	*
Cd	mg kg^{-1}	0.5–2.5	2	*	2.4–12.5	2–4	7
Cr	mg kg^{-1}	60–300	2–3	*	180–5,000	3–4	1,000
Cu	mg kg^{-1}	20–150	2	*	90–1,500	2–4	400
Hg	mg kg^{-1}	0.1–1	1–2	0.6	0.8–5	2–4	*
Ni	mg kg^{-1}	37–130	2–3	*	45–1,500	2–5	*
Pb	mg kg^{-1}	30–120	1–2	*	100–1,500	2–5	*
Zn	mg kg^{-1}	160–700	1–3	*	500–10,000	2–5	*
sum7 PCBs	µg kg^{-1}	2–500	2–3	*	2–1,000	1–4	*
PCB28	µg kg^{-1}	1–25	*	*	6–50	*	*
PCB52	µg kg^{-1}	1–25	*	*	3–50	*	*
PCB101	µg kg^{-1}	2–50	*	*	6–100	*	*
PCB118	µg kg^{-1}	3–25	*	*	10–50	*	*
PCB138	µg kg^{-1}	4–50	*	*	12–100	*	*
PCB153	µg kg^{-1}	4–50	*	*	15–100	*	*
PCB180	µg kg^{-1}	2–25	*	*	6–50	*	*

Notes:

[1] Portugal uses a class range system for assessment of dredged material; numbers in this column show over which classes the target or limit range values fall.

[2] Where maximum concentration range was in the fine fraction (<20 µm), the maximum concentration in the <2 mm is given in this column.

A summary of the preparation and digestion or extraction methods for the assessment of dredged material used by member countries is given in Table A3.16. Most countries analyse the <2 mm fraction; Belgium and Spain examine the <63 µm fraction and Germany the <20 µm. Analysis of metals is done after digestion by HF (total) or HNO_3 (partial). It may thus be difficult to directly compare action level values when differing techniques are applied. Likewise, for organic contaminants different results can be found when soft extractions like Acetone-SPE, or extractions under harsh conditions using ASE or saponification, are applied.

Further notes to Table A3.5

A three-category approach is assumed unless stated below. Tables A3.15–A3.18 show actual concentration data for sediment quality criteria.

BELGIUM

Dredged material to be dumped at sea must satisfy the criteria shown in Table A3.15. If analysis results exceed the limit for three of the criteria, the dredged material may not be dumped at sea. If a concentration falls between the target value and the limit value, the number of sample analyses is increased. If concentrations are still in this zone, then bioassays will be undertaken. http://www.mumm.ac.be/EN/Management/Sea-based/dredging.php.

DENMARK

Denmark is currently developing its system for licensing disposal at sea, and the planned approach is to use locally derived background concentrations. The current view of the Danish EPA is to develop a three-category system according to the following limits. The proposed approach to management of dredged material disposal is as follows:

Decision	Metals	Organics
Disposal at sea permitted	≤ 2× background	≤ 2× background
Disposal subject to conditions	≤ 2× background	40–100 × background
Disposal at sea not permitted	> 2× background	>40 × – >100 × background

For TBT, the Danish EPA may allow sea disposal on the basis of the total amount of substance above the "background" level, if the total amount of TBT to be disposed is less than 1 kg and no acute effects are expected.

Under application of the new regulations, the contaminants which will cause the most problems for dredging applications are TBT > PAHs > Cu and Hg.

FINLAND

A case-by-case system is used for issuing permits for the disposal of dredged material at sea. A project is currently examining sediment quality criteria for dredged material, although the implementation dates are not yet fixed. The proposed criteria have been set according to the following principles:

For metals, Level 1 is usually set at 1.5 × average of background values (2.5 × for mercury), while Level 2 is based on the highest allowable level of toxicity (Maximum Acceptable Risk Level = MAR) or on the Canadian toxicity threshold level (TEL).

For PCBs and DDT, Levels 1 and 2 are set the same as those in the Netherlands. Only the *p,p'* forms of DDT and its derivatives are considered; when measuring accuracy is 0.01 mg kg^{-1}, only threefold concentrations and higher are counted.

For mineral oils, Levels 1 and 2 are set as in the Netherlands from the PAH MAR concentrations. For PAH compounds, Level 1 = MAR/100; Level 2 < MAR. For dioxins and furans, the limit values are taken from the proposal of the National Health Institute, where toxicity is calculated according to the newest WHO TEF values for mammals.

References

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GERMANY

Germany has established action levels for the chemical and biological assessment of dredged material from the German federal waterways. Currently, these action levels do not apply to waters under the responsibility of the Federal States (Länder). However, they are currently under review with the aim of extending their applicability to all German waterways. Dredge disposal quality criteria are derived from presently prevailing contaminant concentrations in the North Sea Wadden Sea sediments from 1982–1987. A three-category approach is used for disposal at sea:

Action Level 1: reference value derived from locally derived sediments (Wadden Sea);

Action Level 2: five times the reference value for metals and three times for organic contaminants.

The action levels represent management values and are neither ecotoxicological quality criteria nor quality targets (Table A3.15). Action levels for organotin compounds are still under discussion.

In order to describe the toxicologically effective pollution load of sediments and dredged materials, an evaluation method for the toxicological investigation of pore water, sediment elutriates, and extracts was developed (Krebs, 1988, 1999). These phases should be examined using standardized tests of aquatic toxicity, employing dilution series in geometric sequence with a dilution factor of two. The toxicological evaluation is then simplified to identifying the first dilution step that is no longer toxic.

For the numerical designation of the toxicity, the first non-toxic dilution factor is used. The pT-value (potentia toxicologiae = toxicological exponent) is the exponent of toxicity and is defined as the negative binary logarithm of the first non-toxic dilution factor in a dilution series in geometric sequence with a dilution factor of two. Accordingly, the pT-value gives an indication of how many times a sample must be diluted in the ratio 1:2 to reach a stage of no toxic effect.

The pT-value of the most sensitive organism within a test battery determines the toxicity class of the dredged material. Different bioassays and different test phases (e.g., pore water, elutriate) are considered equal in rank. The toxicity classes are assigned Roman numerals. If the highest pT-value is 5, for instance, then the tested material is assigned to toxicity class V. The highest level of this classification of dredged material is toxicity class VI, which includes pT-values from 6 to ∞. Regarding the handling of dredged material, the toxicity classes defined by the pT-value method are assigned to the categories “unproblematic” (toxicity classes 0, I, II), “moderately critical” (toxicity classes III and IV), and “highly critical” (V and VI) (Table A3.19).

References

Krebs, F. 1988. Der pT-Wert: Ein gewässer-ökotoxikologischer Klassifizierungsmaßstab (The pT-value: An aquatic ecotoxicological classification index). GIT-Fachzeitschrift für das Laboratorium, 32: 293–296.

Krebs, F. 1999. Ökotoxikologische Klassifizierung von Sedimenten mit Hilfe der pT-Wert-Methode (Ecotoxicological classification of sediments with the pT-value method). *In* Methoden zur Erkundung, Untersuchung und Bewertung von Sedimentablagerungen und Schwebstoffen in Gewässern (Exploration, investigation and assessment of sediment deposits and suspended particulate matter in aquatic systems). Ed. By U. Kern and B. Westrich. Schriftenreihe des Deutschen Verbandes für Wasserwirtschaft und Kulturbau e.V. (DVWK). Heft, 128: 297–303.

ITALY

Legislation involves a number of decrees which have been issued over the past twenty years. The sea disposal of dredged material which can be classified as toxic/noxious is not allowed under Italian legislation (DECRETO 24 gennaio 1996).

THE NETHERLANDS

A maximum concentration limit is set for material for dredged disposal at sea (Table A3.18). There are also lower limits for particular uses of water bodies. A higher limit is set for metals which are known to be in anaerobic sediments, since these will be bound as sulphides.

NORWAY

Dredging activities are of minor importance in Norway (few hundred thousand tonnes annually), and data are currently assessed against the environmental quality standards (Classification of Environmental Quality and Degree of Pollution (CEQDP) in fjords and coastal waters).

A case-by-case basis for the assessment of dredged material is performed. Certain analytes must be measured in material proposed for dredge disposal, while others may be required by the authority according to local conditions. The overall authority is the State Pollution Control Authority, but disposal permits may also be granted by county authorities.

PORTUGAL

Legislation is used to regulate dredged material disposal (DR II n° 141 21/6/1995). The quality criteria standard regulations define the method for disposal of the dredged material. A five-category approach is used to classify material for disposal (Table A3.17), as follows:

Class 1: may be disposed of in the aquatic medium or at places exposed to erosion or used to feed beaches without restrictions.

Class 2: may be disposed of in the aquatic medium subjected to the characteristics of the receiving medium and its legitimate use.

Class 3: may be disposed of on land to reclaim an area to the sea and, if to be disposed at sea, requires adequate study of the disposal place and post-monitoring.

Class 4: to be disposed of on land in a prepared place, including soil impermeability measures and with a recommendation for the spoil to be covered with impermeable soil.

Class 5: should not be dredged. If dredging is absolutely necessary, the dredged material must be treated as industrial waste (specific law being applied). It is forbidden to be disposed of in water or on land.

REPUBLIC OF IRELAND

Figure A3.1 summarizes the Republic of Ireland's existing procedures towards dredged material assessment. To date, the sediment chemistry of dredged material in Ireland has been considered on a case-by-case basis, using provisional action levels. These are guideline figures for the assessment of the sediment and it is intended to derive final action levels for sediment chemistry. These will be built around the background quality of Irish sediments, taking into account the experiences of other countries.

If the volume of sediment is below a provisional figure of 10,000 m^3, and the material is thought to be unaffected by local sources of contamination or is composed entirely of material greater than 2 mm particle size (gravel and larger), then further testing may not be required. This may also be the case for material dredged from capital projects. Material consisting largely of sand is unlikely to be contaminated (apart from flakes of TBT) and a permit may be granted without further testing, providing that no known source of TBT exists in the vicinity.

SPAIN

The Spanish recommendations were published by CEDEX in 1994 and are currently under revision. They are expected to be legally implemented towards the end of 2004. Standards apply to fine sediments (<63 µm) with more than 10% of fine material and to those with more than 10% of organic matter using a three-category approach. Sediments with metal concentrations above the second level and = 8 × Level 2 have to be isolated and belong to category IIIa. Sediments with metal concentrations higher than 8 × level 2 must be isolated into containers or into a contained area (category IIIb). Locally derived background concentrations are under development.

Bioassays under development for application to dredged material assessment are Microtox, *Ampelisca* or *Corophium* and a crustacean.

UNITED KINGDOM

England and Wales, Scotland, and Northern Ireland have separate but equivalent bodies that implement marine disposal of dredged material. A case-by-case approach is adopted throughout the UK, although England and Wales are currently developing an action level approach towards the assessment of dredged material. Dredging disposal to sea is governed by Part II of the Food and Environmental Protection Act 1985 (FEPA). Assessments of the impacts are based on:

- the material to be disposed (physico-chemical analysis);
- method of dredging and disposal;
- disposal site characteristics;
- other disposal options, especially beneficial uses.

The basis for setting standards was suggested in two Environmental Quality Objectives relevant to sediments (MAFF, 1989):

<u>1 General ecosystem conservation</u>

Objective: Maintenance of environmental quality so as to protect aquatic life and dependent non-aquatic organisms, such that the ecosystem is typical of coastal water with those physical characteristics and latitude. Basis of standard: Grain-size, carbon/nitrogen and toxic substances to be below levels of effect, and within any EQS set by relevant legislation.

<u>2 Preservation of the natural environment</u>

Objective: Outside of the immediate disposal zone, the quality of the receiving environment will be indistinguishable from that of the adjacent estuarine or marine environment. Basis of standard: Minimal percentage change over background levels of metals and other contaminants. No continuing upward trends after "steady-state" is achieved.

Action levels currently being derived for England and Wales have been set according to the following approach. Action Level 1 is set at an estimate of the median background concentration, i.e., based on sediment chemistry. Action level 2 is guided by the ERM and TEL values produced by Long *et al.* (1995, 1998) and Long and MacDonald (1998). It is expected that experience in the use of the values, and collection of data applicable to the local situation, will allow revision of the Action Levels in a few years' time.

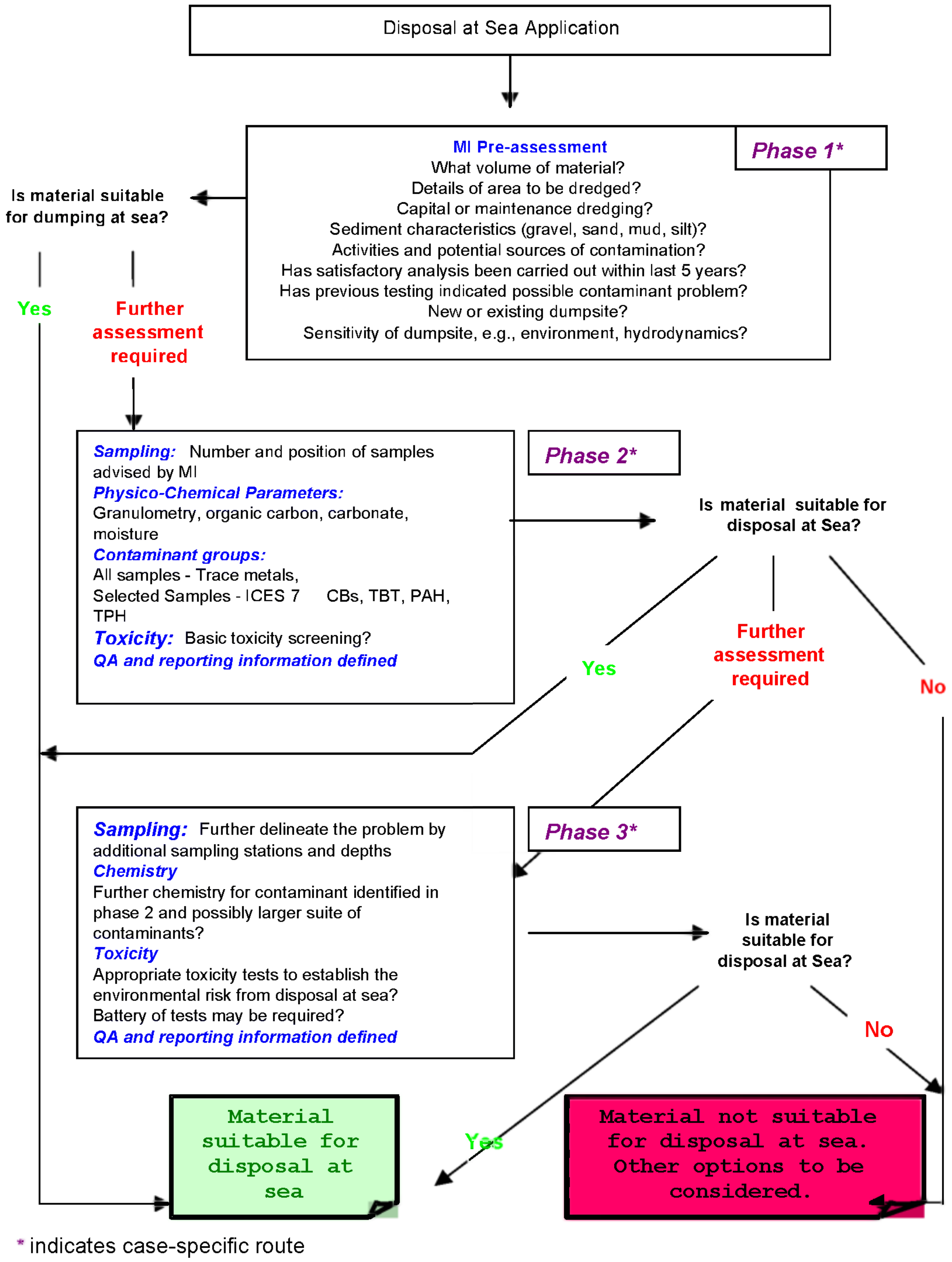

Figure A3.1. Assessment of sediments with respect to chemical contamination for disposal at sea permission in the Republic of Ireland (with permission from Margot Cronin and Evin McGovern, Marine Institute).

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TABLES A3.6–A3.14 ENVIRONMENTAL QUALITY STANDARDS FOR SEDIMENTS

Table A3.6. Canadian sediment quality guidelines for the protection of aquatic life. Interim marine sediment quality guidelines (ISQGs; dry weight), probable effects levels (PELs; dry weight), and incidence (%) of adverse biological effects in concentration ranges defined by these values.

Substance	units	ISQG	PEL	% ≤ ISQG	ISQG < %PEL	% ≥ PEL
As	mg kg^{-1}	7.24	41.6	3	13	47
Cd	mg kg^{-1}	0.7	4.2	6	20	71
Cr	mg kg^{-1}	2.26	4.79	9	12	17
Cu	mg kg^{-1}	18.7	108	9	22	56
Hg	mg kg^{-1}	0.13	0.7	8	24	37
Pb	mg kg^{-1}	30.2	112	6	26	58
Zn	mg kg^{-1}	124	271	4	27	65
DDT[1]	µg kg^{-1}	1.19	4.77	8	5	59
DDD[1]	µg kg^{-1}	1.22	7.81	4	11	46
DDE[1]	µg kg^{-1}	2.07	374	5	16	50
Dieldrin	µg kg^{-1}	0.71	4.3	4	13	50
Total PCBs	µg kg^{-1}	21.5	189	16	37	55
PCDDs and PCDFs	ng TEQ kg^{-1}	0.85	21.5	*	*	*
Acenaphthene	µg kg^{-1}	6.71	88.9	8	29	57
Acenaphthylene	µg kg^{-1}	5.87	128	7	14	51
Anthracene	µg kg^{-1}	46.9	245	9	20	75
Benz[*a*]anthracene	µg kg^{-1}	74.8	693	9	16	78
Benzo[*a*]pyrene	µg kg^{-1}	88.8	763	8	22	71
Chrysene	µg kg^{-1}	108	846	9	19	72
Dibenz[*a,h*]anthracene	µg kg^{-1}	6.22	135	16	12	65
Fluoranthene	µg kg^{-1}	113	1494	10	20	80
Fluorene	µg kg^{-1}	21.2	144	12	20	70
2-Methylnaphthalene	µg kg^{-1}	20.2	201	0	23	82
Naphthalene	µg kg^{-1}	34.6	391	3	19	71
Phenanthrene	µg kg^{-1}	86.7	544	8	23	78
Pyrene	µg kg^{-1}	153	1398	7	19	83

Notes:
[1] sum of *p,p'* and *o,p'* isomers

Table A3.7. Environmental quality classifications for sediments in The Netherlands. All values dry weight sediment. Values normalized to sediment content 5% TOC, 10% clay for metals and 5% organic carbon for organic contaminants.

		Long-term	**Short-term**
Contaminant	**Unit**	**NC**	**MPC**
Cd	mg kg^{-1}	0.8	12
Inorganic Hg	mg kg^{-1}	0.3	10
Organic Hg	mg kg^{-1}	0.3	1.4
Cu	mg kg^{-1}	36	73
Ni	mg kg^{-1}	35	44
Pb	mg kg^{-1}	85	530
Zn	mg kg^{-1}	140	520
Cr	mg kg^{-1}	100	380
As	mg kg^{-1}	29	55
Naphthalene	µg kg^{-1}	1	100
Anthracene	µg kg^{-1}	1	100
Phenanthrene	µg kg^{-1}	5	500
Fluoranthene	µg kg^{-1}	30	3,000
Benz[*a*]anthracene	µg kg^{-1}	3	400
Chrysene	µg kg^{-1}	100	11,000
Benzo[*k*]fluoranthene	µg kg^{-1}	20	2,000
Benzo[*a*]pyrene	µg kg^{-1}	3	3,000
Benzo[*ghi*]perylene	µg kg^{-1}	80	8,000
Indeno[1,2,3*cd*]pyrene	µg kg^{-1}	60	6,000
Pentachlorobenzene	µg kg^{-1}	1	100
Hexachlorobenzene	µg kg^{-1}	0.05	5
Pentachlorophenol	µg kg^{-1}	2	300
Aldrin	µg kg^{-1}	0.06	6
Dieldrin	µg kg^{-1}	0.5	450
Endrin	µg kg^{-1}	0.04	4
DDT	µg kg^{-1}	0.09	9
DDD	µg kg^{-1}	0.02	2
DDE	µg kg^{-1}	0.01	1
α -Endosulfan	µg kg^{-1}	0.01	1
α-HCH	µg kg^{-1}	3	290
β –HCH	µg kg^{-1}	9	920
γ-HCH	µg kg^{-1}	0.05	230
Heptachlor	µg kg^{-1}	0.7	0.7
Heptachlorepoxide	µg kg^{-1}	0.0002	0.02
Chlordane	µg kg^{-1}	0.03	3
Total Hydrocarbons	µg kg^{-1}	50	1,000
PCB28	µg kg^{-1}	1	4
PCB52	µg kg^{-1}	1	4
PCB101	µg kg^{-1}	4	4
PCB118	µg kg^{-1}	4	4
PCB138	µg kg^{-1}	4	4
PCB153	µg kg^{-1}	4	4
PCB180	µg kg^{-1}	4	4

Table A3.8. Environmental quality classifications for sediment in Norway, < 2mm fraction, dry weight.

Contaminant	Units	Class 1 (Good)	Class 2 (Fair)	Class 3 (Poor)	Class 4 (Bad)	Class 5 (Very bad)
Ag	mg kg^{-1}	<0.3	0−1.3	1–5	5−10	>10
As	mg kg^{-1}	<20	20–80	80–400	400−1,000	>1,000
Cd	mg kg^{-1}	<0.25	0.25−1	1–5	5−10	>10
Cr	mg kg^{-1}	<70	70–300	300−1,500	1,500-5,000	>5,000
Cu	mg kg^{-1}	<35	35−150	150–700	700−1,500	>1,500
F	mg kg^{-1}	<800	800–3,000	3,000–8,000	8,000–20,000	>20,000
Hg	mg kg^{-1}	<0.15	0.15–0.6	0–3	3–5	>5
Pb	mg kg^{-1}	<30	30−120	120–600	600−1,500	>1,500
Ni	mg kg^{-1}	<30	30−130	130–600	600−1,500	>1,500
Zn	mg kg^{-1}	<150	150–650	650–3,000	3,000−10,000	>10,000
TBT[1]	µg kg^{-1}	<1	5	5–20	20−100	>100
Sum PAHs[2]	µg kg^{-1}	<300	300–2,000	2,000–6,000	6,000–20,000	>20,000
Sum 7 PCBs[3]	µg kg^{-1}	<5	5–25	25−100	100–300	>300
B[*a*]P	µg kg^{-1}	<10	10–50	50–200	200–500	>500
EPOCl[4]	µg kg^{-1}	<100	100–500	500–2,000	2,000−15,000	>15,000
$TE_{PCDF/D}$[5]	µg kg^{-1}	<0.01	0.01–0.03	0.03–0.10	0.10–0.5	>0.5
Sum DDT[6]	µg kg^{-1}	<0.5	0.5–2.5	2.5−10	10–50	>50
Hexachlorobenzene	µg kg^{-1}	<0.5	0.5–2.5	2.5−10	10–50	>50

Notes:

[1] TBT: Tributyltin as the molecule.

[2] PAHs: Polycyclic aromatic hydrocarbons, sum PAHs: sum of tri- to hexacyclic compounds including the sixteen in EPA protocol 8310. The dicyclic naphthalenes are not included. The sum also includes all potentially carcinogenic PAHs (group 2A and 2B in IARC, 1987).

[3] PCBs: Polychlorinated biphenyls, sum PCB-7: total of congeners (CB) 28, 52, 101, 118, 138, 153, 180.

[4] EPOCl: Extractable persistent organically bound chlorine.

[5] $TE_{PCDF/D}$: Toxic Equivalents of polychlorinated dibenzo-*p*-dioxins and dibenzofurans (dioxins). Within the PCDD/PCDF there is a small group of very toxic compounds. These can be expressed as toxic-equivalents (TE) of the most toxic compound (2,3,7,8-TCDD).

[6] DDT: Diphenyltrichloroethylene: sum DDT is the total of DDT and metabolites DDE and DDD.

Table A3.9. Swedish Environmental Criteria for organic micro-pollutants in coastal and marine sediments, measured in the <2 mm fraction and expressed in µg kg^{-1} dry matter.

Substance	Class I "None"	Class II "Low concentration"	Class III "Moderate concentration"	Class IV "High concentration"	Class V "Very high concentration"
Phenanthrene	0	0 – 10	10 – 30	30 – 100	> 100
Anthracene	0	0 – 2	2 – 8	8 – 30	> 30
Fluoranthene	0	0 – 20	20 – 80	80 – 270	> 270
Pyrene	0	0 – 12	12 – 50	50 – 200	> 200
Benzo[*a*]anthracene	0	0 – 10	10 – 35	35 – 110	> 110
Chrysene	0	0 – 13	13 – 50	50 – 180	> 180
Benzo[*b*]fluoranthene	0	0 – 50	50 – 150	150 – 400	> 400
Benzo[*k*]fluoranthene	0	0 – 20	20 – 50	50 –160	> 160
Benzo[*a*]pyrene	0	0 – 20	20 – 60	60 – 180	> 180
Benzo[*ghi*]perylene	0	0 – 30	30 – 100	100 – 350	> 350
Indeno[*cd*]pyrene	0	0 – 50	50 – 170	170 – 600	> 600
Sum 11 PAHs	0	0 – 280	280 – 800	800 – 2,500	> 2,500
HCB	0	0 – 0.04	0.04 – 0.2	0.2 – 1	> 1
PCB28	0	0 – 0.06	0.06 – 0.2	0.2 – 0.6	> 0.6
PCB52	0	0 – 0.06	0.06 – 0.2	0.2 – 0.8	> 0.8
PCB101	0	0 – 0.16	0.16 – 0.6	0.6 – 2	> 2
PCB118	0	0 – 0.15	0.15 – 0.6	0.6 – 2	> 2
PCB153	0	0 – 0.03	0.03 – 0.3	0.3 – 3.5	> 3.5
PCB138	0	0 – 0.3	0.3 – 1.2	1.2 – 4.1	> 4.1
PCB180	0	0 – 0.1	0.1 – 0.4	0.4 – 1.9	> 1.9
Sum 7 PCBs (Dutch)	0	0 – 1.3	1.3 – 4	4 – 15	> 15
Total PCBs	0	0 – 5	5 – 20	20 – 75	> 75
α-HCH	0	0 – 0.01	0.01 – 0.07	0.07 – 0.3	> 0.3
β-HCH	0	0 – 0.03	0.03 – 0.3	0.3 – 3	> 3
γ-HCH	0	0 – 0.01	0.01 – 0.1	0.1 – 1.3	> 1.3
Sum HCHs	0	0 – 0.03	0.03 – 0.3	0.3 – 3	> 3
γ-chlordane	0	0 – 0.01	0.01 – 0.04	0.04 – 0.1	> 0.1
α-chlordane	0	0 – 0.02	0.02 – 0.04	0.04 – 0.1	> 0.1
Trans-nonachlor	0	0 – 0.02	0.02 – 0.05	0.05 – 0.15	> 0.15
Sum chlordanes	0	0 – 0.02	0.02 – 0.08	0.08 – 0.3	> 0.3
p,p´-DDT	0	0 – 0.02	0.02 – 0.1	0.1 – 0.7	> 0.7
p,p´-DDE	0	0 – 0.2	0.2 – 0.7	0.7 – 2.5	> 2.5
p,p´-DDD	0	0 – 0.13	0.13 – 0.8	0.8 – 5	> 5
Sum DDT	0	0 – 0.2	0.2 – 1	1 – 6	> 6
EOCl	0	0 – 600	600 – 4,000	4,000 – 30,000	> 30,000
EOBr	0	0 – 400	400 – 1,000	1,000 – 3,000	> 3,000
EPOCl	0	0 – 150	150 – 700	700 – 3,000	> 3,000
EPOBr	0	0 – 90	90 – 250	250 – 800	> 800

Notes:
EOCl = Total extractable organic chlorine; EPOCl = Total persistent extractable organic chlorine;
EOBr = Total extractable organic bromine; EPOBr = Total persistent extractable organic bromine.

Class 1 is set at zero as these contaminants do not occur naturally. The boundary between classes 2 and 3 is at the 5th percentile of the data set, corresponding to samples not affected by point sources. The boundary between classes 3 and 4 was determined from the 5th percentile multiplied by the square root of the ratio between the 5th and 95th percentiles. The 95th percentile has been used for the boundary between classes 4 and 5, except in the case of EOCl (90th percentile).

Table A3.10. Background values for metals in sediment (mg kg^{-1} dry weight) used in the Swedish classification system of Environmental Quality Criteria.

Method	**As**	**Cd**	**Co**	**Cr**	**Cu**	**Hg**	**Ni**	**Pb**	**Zn**
Partial digestion (7 M HNO_3)	10	0.2	12	40	15	0.04	30	25	85
Total digestion	10	0.2	14	80	15	0.04	33	31	85

Table A3.11. The classification factors of the deviation in the surficial sediment (0–2 cm) from the background value, based on partial digestion of the sediment (leaching at 7 M HNO_3).

Metal Partial digestion (7 M HNO_3)	**Class I "No/very small deviation"**	**Class II "Small deviation"**	**Class III "Clear deviation"**	**Class IV "Large deviation"**	**Class V "Very large deviation"**
Arsenic	≤ 1.0	1.0 – 1.7	1.7 – 2.8	2.8 – 4.5	> 4.5
Cadmium	≤ 1.0	1.0 – 2.5	2.5 – 6.0	6.0 – 15	> 15
Cobolt	≤ 1.0	1.0 – 1.7	1.7 – 2.9	2.9 – 5.0	> 5.0
Chromium	≤ 1.0	1.0 – 1.2	1.2 – 1.5	1.5 – 1.8	> 1.8
Copper	≤ 1.0	1.0 – 2.0	2.0 – 3.3	3.3 – 5.3	> 5.3
Mercury	≤ 1.0	1.0 – 3.0	3.0 – 10	10 – 25	> 25
Nickel	≤ 1.0	1.0 – 1.5	1.5 – 2.2	2.2 – 3.3	> 3.3
Lead	≤ 1.0	1.0 – 1.6	1.6 – 2.6	2.6 - 4.4	> 4.4
Zinc	≤ 1.0	1.0 – 1.5	1.5 – 2.4	2.4 – 4.2	> 4.2

Table A3.12. The classification factors of the deviation in the surficial sediment (0–2 cm) from the background value, based on total digestion of the sediment.

Metal Total digestion	**Class I "No/very small deviation"**	**Class II "Small deviation"**	**Class III "Clear deviation"**	**Class IV "Large deviation"**	**Class V "Very large deviation"**
Arsenic	≤ 1.0	1.0 – 1.6	1.6 – 2.6	2.6 – 4.0	> 4.0
Cadmium	≤ 1.0	1.0 – 2.5	2.5 – 6.0	6.0 – 15	> 15
Cobalt	≤ 1.0	1.0 – 1.4	1.4 – 2.0	2.0 – 2.8	> 2.8
Chromium	≤ 1.0	1.0 – 1.4	1.4 – 2.0	2.0 – 2.8	> 2.8
Copper	≤ 1.0	1.0 – 2.0	2.0 – 4.0	4.0 – 8.0	> 8.0
Mercury	≤ 1.0	1.0 – 2.6	2.6 – 6.8	6.8 – 18	> 18
Nickel	≤ 1.0	1.0 – 1.3	1.3 – 1.7	1.7 – 2.4	> 2.4
Lead	≤ 1.0	1.0 – 1.5	1.5 – 2.2	2.2 – 3.3	> 3.3
Zinc	≤ 1.0	1.0 – 1.5	1.5 – 2.3	2.3 – 3.5	> 3.5

Table A3.13. The actual concentration boundaries of the Swedish classification system of superficial sediment (0–2 cm), based on partial digestion of the sediment. The values have been calculated by using the background value (Table A3.10) and the factors in Table A3.11.

Metal Partial digestion (7 M HNO_3)	**Class I "No/very small deviation"**	**Class II "Small Deviation"**	**Class III "Clear deviation"**	**Class IV "Large deviation"**	**Class V "Very large deviation"**
Arsenic	≤ 10	10 – 17	17 – 28	28 – 45	> 45
Cadmium	≤ 0.2	0.2 – 0.5	0.5 – 1.2	1.2 – 3	> 3
Cobalt	≤ 12	12 – 20	20 – 35	35 – 60	> 60
Chromium	≤ 40	40 – 48	48 – 60	60 – 72	> 72
Copper	≤ 15	15 – 30	30 – 50	50 – 80	> 80
Mercury	≤ 0.04	0.04 – 0.12	0.12 – 0.4	0.4 – 1.0	> 1.0
Nickel	≤ 30	30 – 45	45 – 66	66 – 99	> 99
Lead	≤ 25	25 – 40	40 – 65	65 – 110	> 110
Zinc	≤ 85	85 – 128	128 – 204	204 – 357	> 357

Table A3.14. The actual concentration boundaries of the Swedish classification system of superficial sediment (0–2 cm), based on total digestion of the sediment. The values have been calculated by using the background value (Table A3.10) and the factors in Table A3.12.

Metal Total digestion	**Class I "No/Very small deviation"**	**Class II "Small deviation"**	**Class III "Clear deviation"**	**Class IV "Large deviation"**	**Class V "Very large deviation"**
Arsenic	≤ 10	10 – 16	16 – 26	26 – 40	> 40
Cadmium	≤ 0.2	0.2 – 0.5	0.5 – 1.2	1.2 – 3	> 3
Cobalt	≤ 14	14 – 20	20 – 28	28 – 40	> 40
Chromium	≤ 80	80 – 112	112 – 160	160 – 224	> 224
Copper	≤ 15	15 – 30	30 – 60	60 – 120	> 120
Mercury	≤ 0.04	0.04 – 0.1	0.1 – 0.27	0.27 – 0.72	> 0.72
Nickel	≤ 33	33 – 43	43 – 56	56 – 79	> 79
Lead	≤ 31	31 – 47	47 – 68	68 – 102	> 102
Zinc	≤ 85	85 – 128	128 – 196	196 – 298	> 298

Notes to Tables A3.13 and A3.14:
Class 1 is derived from the background value (Table A3.10). The other class boundaries have been defined statistically by multiplying each value of the boundary with a factor. This factor is defined as the cubic root of the ratio between the boundary between classes 4 and 5 and the background value. The boundary between the classes 2 and 3 is found when multiplying the factor with the background value. The boundary between the classes 3 and 4 is calculated from the value of the class boundary by multiplying this value with the factor. The boundary between classes 4 and 5 is given by the 95th percentile of the partial digestion result and the 99th percentile for the total digestion. Class 5 is further divided into three subclasses focused on finding contaminated "hot spots" that have to be treated in one or the other way to diminish the health risk.

TABLES A3.15–A3.19 STANDARDS FOR DREDGED MATERIAL DISPOSAL AT SEA

Table A3.15. Action levels for dredged material disposal assessment, data based on dry weight.

	Grain size/ sediment composition	Belgium < 2 mm		Finland (proposed) 25% clay 10% organics		France < 2 mm		Germany < 20 µm		Norway < 2 mm		Spain < 63 µm >10% fine fraction and >10% organic matter		Sweden < 2 mm		United Kingdom England+Wales[1] < 63 µm Action levels being derived	
Contaminant	Action level	1	2	1	2	1	2	1	2	1	2	1	2	1	2	1	2
As	mg kg^{-1}	20	100	15	60	25	50	30	150	80	1,000	80	200	30	100		
Cd	mg kg^{-1}	2.5	7	0.5	2.5	1.2	2.4	2.5	12.5	1	10	1	5	0.9	3		
Cr	mg kg^{-1}	60	220	65	270	90	180	150	750	300	5,000	200	1,000	60	200		
Cu	mg kg^{-1}	20	100	50	90	45	90	40	200	150	1,500	100	400	60	200		
Hg	mg kg^{-1}	0.3	1.5	0.1	1	0.4	0.8	1	5	0.6	5	0.6	3	0.3	1		
Ni	mg kg^{-1}	70	280	45	60	37	74	50	250	130	1,500	100	400	45	150		
Pb	mg kg^{-1}	70	350	40	200	100	200	100	500	120	1,500	120	600	30	100		
Zn	mg kg^{-1}	160	500	170	500	276	552	350	1,750	700	10,000	500	3,000	375	1,250		
Total hydrocarbons	mg kg^{-1}	14	36					300	1,000								
Mineral oil	mg kg^{-1}			50	1,500												
Hexachlorobenzene	µg kg^{-1}							2	6								
Sum-DDT[2]	µg kg^{-1}			2.5	30												
DDT	µg kg^{-1}							1	3								
DDE	µg kg^{-1}							1	3								
DDD	µg kg^{-1}							3	10								
Sum 6 PAH[3]	µg kg^{-1}							1	3								
Naphthalene	µg kg^{-1}	1.4–3.5	3.6–9	20	1,000												
Phenanthrene	µg kg^{-1}			50	1,000												
Anthracene	µg kg^{-1}			50	1,000												
Fluoranthene	µg kg^{-1}			20	1,500												
Chrysene	µg kg^{-1}			20	1,000												
Benzo[*a*]anthracene	µg kg^{-1}			20	1,000												
Benzo[*a*]pyrene	µg kg^{-1}			30	1,000			0.05	0.5								
Benzo[*k*]fluoranthene	µg kg^{-1}			30	1,000												
Ideno[*1,2,3cd*]pyrene	µg kg^{-1}			30	1,000												
Benzo[*ghi*]perylene	µg kg^{-1}			20	1,000												
Sum 7 PCBs	µg kg^{-1}	2	2			500	1,000	20	60	25	300	30	100				
PCB28	µg kg^{-1}			1	30	25	50	2	6								
PCB52	µg kg^{-1}			1	30	25	50	1	3								
PCB101	µg kg^{-1}			4	30	50	100	2	6								
PCB118	µg kg^{-1}			4	30	25	50	3	10								
PCB138	µg kg^{-1}			4	30	50	100	4	12								
PCB153	µg kg^{-1}			4	30	50	100	5	15								
PCB180	µg kg^{-1}			4	30	25	50	2	6								
TBT	µg kg^{-1}	3	7	(3)[4]	(200)[4]												
PCDD/F	WHO TEQ ng kg^{-1}			20	500												

Notes: [1] Data for Action Levels for England and Wales are currently being reviewed internally. They should be available shortly. [2] Sum DDT = DDT+DDD+DDE. [3] Sum 6 PAH = fluoranthene + benzo[*b*]fluoranthene + benzo[*k*]fluoranthene + benzo[*a*]pyrene + benzo[*g,h,i*]perylene + indeno[1,2,3-*c,d*]pyrene. [4] In Finland, TBT values are provisional and based on organic tin.

Table A3.16. Summary of preparation, digestion, and extraction methods for the assessment of dredged material (based on OSPAR SEABED 02/2/6 Rev.1-E(L)).

	Belgium	Canada	Denmark	England and Wales	France	Ireland	Netherlands	Norway	Northern Ireland	Portugal	Scotland[1]	Spain	Sweden
Size fraction	< 63 µm		<2 mm	< 2 mm	< 2 mm	< 2 mm	< 2 mm	< 2 mm	<2 mm	< 2 mm	< 2 mm	< 63 µm	
Metals	Total digestion		Total or partial[2]	Partial HNO_3	Total digestion	Total (HF)[3]	Partial (NVN 5770)	Total (HF)[3]	Total (HF)	Total	Partial HNO_3	Partial HNO_3	
PAHs	ASE (high dw); Soxhlet (low dw)			Alkaline saponification	Hexane/ CH_2Cl_2		Acetone-SPE extraction	Soxhlet w/ dichloromethane	Alkaline saponification	Alkaline saponification	Sonication	Sonication cycle hexane/ dichloromethane	
PCBs				Solvent extraction		Soxhlet/ Solvent extraction		Acetone/hexane	Hexane	Soxhlet/ Solvent extraction	Soxhlet	Sonication cycle hexane/ dichloromethane	
HCH				Solvent extraction		Soxhlet/ Solvent extraction		Acetone/hexane	Hexane	Soxhlet/ Solvent extraction	Soxhlet		
DDTs				Solvent extraction		Soxhlet/ Solvent extraction		Acetone/hexane	Hexane	Soxhlet/ Solvent extraction	Soxhlet		
Organotins	Adding of acid, extraction with hexane/ tropolone			Methanol hydroxide					*Iso*-octane (EA)		Soxhlet	Derivation Grignard	

[1] Scotland: samples not sieved prior to determination of organics.
[2] Denmark: total or partial digestion of sediments determined at a local (county) level.
[3] Norway and Ireland: Hg determined after nitric acid digestion.

Table A3.17. Dredged material classification criteria in Portugal. All concentrations in the <2 mm size fraction.

<2mm fraction	Units	Class 1 (clean)	Class 2 (slightly contaminated)	Class 3 (moderately contaminated)	Class 4 (contaminated)	Class 5 (very contaminated)
As	mg kg^{-1}	<20	20–50	50–100	100–500	>500
Cd	mg kg^{-1}	<1	1–3	3–5	5–10	>10
Cr	mg kg^{-1}	<50	50–100	100–400	400–1,000	>1,000
Cu	mg kg^{-1}	<35	35–150	150–300	300–500	>500
Hg	mg kg^{-1}	<0.5	0.5–1.5	1.5–3.0	3.0–10	>10
Pb	mg kg^{-1}	<50	50–150	150–500	500–1,000	>1,000
Ni	mg kg^{-1}	<30	30–75	75–125	125–250	>250
Zn	mg kg^{-1}	<100	100–600	600–1,500	1,500–5,000	>5,000
Sum PCBs	µg kg^{-1}	<5	5–25	25–100	100–300	>300
Sum PAHs	µg kg^{-1}	300	300–2,000	2,000–6,000	6,000–20,000	>20,000
HCB	µg kg^{-1}	<0.5	0.5–2.5	2.5–10	10–50	>50

Table A3.18. Dredged material assessment criteria in The Netherlands. Concentrations in standard sediment consisting of 25% clay (<2 µm) and 10% organic matter (1.7–2 × organic carbon).

	Parameter	Dredged material criteria CTT	Unit
Biological effects test	**Crustacean (*C. volutator*)**	**35**	**Mortality %**
	Microtox SP (*V. fisheri*)	**100**	**TU with grain size corrections**
	DR-CALUX	**50**	**TCDD TEQ pg g^{-1} dw**
Contaminants	Arsenic	29	mg kg^{-1}
	Cadmium	4	mg kg^{-1}
	Chromium	120	mg kg^{-1}
	Copper	60	mg kg^{-1}
	Mercury	1.2	mg kg^{-1}
	Nickel	45	mg kg^{-1}
	Lead	110	mg kg^{-1}
	Zinc	365	mg kg^{-1}
	PAH total	8	mg kg^{-1}
	Mineral oil	1,250	mg kg^{-1}
	Sum 7 PCBs	100	µg kg^{-1}
	Sum DDT	20	µg kg^{-1}
	HCB	20	µg kg^{-1}

Table A3.19. Toxicity classes and sediment characterization for dredged material in Germany (ecotoxicological measurements are made in pore water and eluates).

Strongest dilution without effect	Dilution factor	PT value	Toxicity class	Characterization		Classification
			7-scale system	4-scale assessment	Description	
Original sample	2^{0}	0	0	0	Unproblematic	Category 1
1:2	2^{-1}	1	I	I		
1:4	2^{-2}	2	II	II		
1:8	2^{-3}	3	III	III	Moderately critical	Category 2
1:16	2^{-4}	4	IV	IV		
1:32	2^{-5}	5	V	V	Highly critical	Category 3
≤ (1:64)	$\leq 2^{-6}$	≥ 6	VI	VI		

ANNEX 4: A PRECAUTIONARY APPROACH TO TESTING FOR COMPLIANCE WITH ENVIRONMENTAL OBJECTIVES

1 Introduction

Environmental objectives are often stated with no guidance as to how they should be assessed. For example, the declared OSPAR policy objective for hazardous substances is

"*... achieving concentrations in the environment near background values for naturally occurring substances and close to zero for man-made synthetic substances*" (OSPAR, 1998). A similar objective exists for radioactive substances.

A problem with this objective is the ambiguity associated with the phrases *near background* and *close to zero*. It is also unclear how in practice one might test whether the objective has been met.

In Section 2, a simple statistical framework is described that clarifies objectives about background concentrations and leads to a practical testing procedure for background and other environmental reference values. Section 3 contains a brief discussion of how these reference values should be derived. It is argued that reference values are only practical if supported by an appropriate monitoring programme, and that their choice will involve some balance of environmental and monitoring costs. In Section 4, it is demonstrated how the statistical framework can be applied within the OSPAR Coordinated Environmental Monitoring Programme (CEMP) using data on mercury in plaice sampled at Liverpool Bay, UK. Throughout, *background concentration* is used to denote both background and zero concentrations, and *reference value* is used as a general term for any environmental reference point.

To be specific, this annex will refer to contaminants in the environment, and assume that the environmental policy is for these to be reduced. However, the discussion could apply to any biological, ecological, biochemical, or chemical response.

2 Statistical Framework for Testing for Compliance with a Reference Value

2.1 Brown and green statistical tests

We assume there is an estimate $\hat{\mu}_T$ of μ_T, the mean contaminant concentration in the assessment year T. (Often $\hat{\mu}_T$ will be estimated from a series of annual indices of contaminant concentration $y_1, y_2, \ldots, y_T$.) Further, we assume that $\hat{\mu}_T$ has variance

$$Var(\hat{\mu}_T) = \sigma_\mu^2$$

estimated by $\hat{\sigma}_\mu^2$ on *df* degrees of freedom.

The traditional formulation of a test of $\hat{\mu}_T$ against a general reference value R is to postulate a null hypothesis representing the *status quo* of the form

$$H_0: \mu_T \le R$$

and reject this in favour of an alternative hypothesis

$$H_1: \mu_T > R$$

if there is sufficient evidence against H_0.

An appropriate statistical test is based on the lower $100(1-\alpha)\%$ confidence limit for μ_T,

$$LCL_T = \hat{\mu}_T + t_{df}(\alpha) \times \hat{\sigma}_\mu$$

where α is the size of the test and $t_{df}(\alpha)$ is the α–quantile of a Student's t-distribution with *df* degrees of freedom. Specifically, we would reject H_0 in favour of H_1 if

$$LCL_T > R.$$

For simplicity, we label the decision to accept H_0 as *Uncontaminated*, and the decision to accept H_1 as *Contaminated*.

This test is sometimes referred to as a *Brown* test (cf., Fox, 2001), since the benefit of doubt is against the environment. The implications of this are seen in the stylized Figure A4.1a, which shows the probability of concluding *Uncontaminated* as the true mean varies. The solid line corresponds to a satisfactory level of precision in the monitoring programme, in the sense that the probability of concluding *Uncontaminated* is less than 10% when the mean is above the point labelled *Above R*. The test is non-precautionary because there is a greater than 10% chance of concluding *Uncontaminated* when the mean concentration is between R and *Above R*.

A danger with non-precautionary tests is demonstrated by the broken line in Figure A4.1a, which corresponds to poor monitoring precision. Now the probability of concluding *Uncontaminated* is still more than 50% even when the true mean concentration has reached the point labelled *Well above R*.

Brown tests are counter to the precautionary principle advocated by OSPAR (OSPAR, 1998). A precautionary, or *Green* test, is obtained by reversing the roles of the hypotheses, redefining the *status quo* as

H_0: $\mu_T > R$

and rejecting this in favour of

H_1: $\mu_T \leq R$

if there is sufficient evidence against H_0.

An appropriate statistical test is based on the upper 100(1−α)% confidence limit

$$LCL_T = \hat{\mu}_T + t_{df}(1-\alpha) \times \hat{\sigma}_\mu$$

and H_0 would be rejected in favour of H_1 (i.e., conclude *Uncontaminated*) if

$UCL_T \leq R$.

The properties of this test are demonstrated in Figure A4.1b. Here the probability of wrongly concluding *Uncontaminated* is below α = 5% for both satisfactory and imprecise monitoring. The burden of demonstrating compliance with R now rests with the monitoring programme. With satisfactory precision, the probability of concluding *Uncontaminated* is better than 90% when μ_T is at the point labelled *Below R*, but with imprecise monitoring the probability is less than 50% even when μ_T is at the point labelled *Well below R*.

2.2 Green tests for compliance with a background concentration

Unfortunately, the simple green test described above does not work when the reference is a background concentration, B. Figure A4.2a shows the properties of a green test where

H_0: $\mu_T > B$

is rejected in favour of

H_1: $\mu_T \leq B$

when

$UCL_T \leq B$.

Since, by definition, the true mean concentration cannot fall below B, the maximum probability of concluding *Uncontaminated* is 5% and occurs when $\mu_T = B$. The broken line corresponds to the unachievable probabilities when $\mu_T < B$.

However, Figure A4.2a also shows a modified green test, corresponding to

H_0: $\mu_T > kB$

H_1: $\mu_T \leq kB$

where $k > 1$, and H_0 is rejected in favour of H_1 when

$UCL_T \leq kB$.

With this modified test, the probability curve has been shifted to the right, so that the probability of concluding *Uncontaminated* is 90% when $\mu_T = B$, with a risk no greater than 5% of concluding *Uncontaminated* when $\mu_T \geq kB$.

A simple, precautionary method for testing for compliance with a background concentration is therefore possible if we can identify a *Near-background* reference concentration below which the mean concentration is considered to be "sufficiently near to background". We can then construct a statistical test of the null and alternative hypotheses

H_0: $\mu_T >$ Near-background concentration

H_1: $\mu_T \leq$ Near-background concentration

and conclude *Uncontaminated* when

$UCL_T \leq$ Near-background concentration.

Figure A4.2b shows the properties of this test. It again demonstrates the high level of environmental protection when monitoring is imprecise. However, when monitoring is very precise, the probability of concluding *Uncontaminated* approaches unity when mean concentrations are above B but below *Near B* (i.e., sufficiently near to background). This reflects the trade-off between accepting a tolerable level of contamination and the benefits of having a precautionary test.

2.3 Green tests for compliance with an upper reference concentration

The modified green test is not necessary when testing for compliance with an upper reference concentration. However, in some contexts it may be appropriate. The important question is:

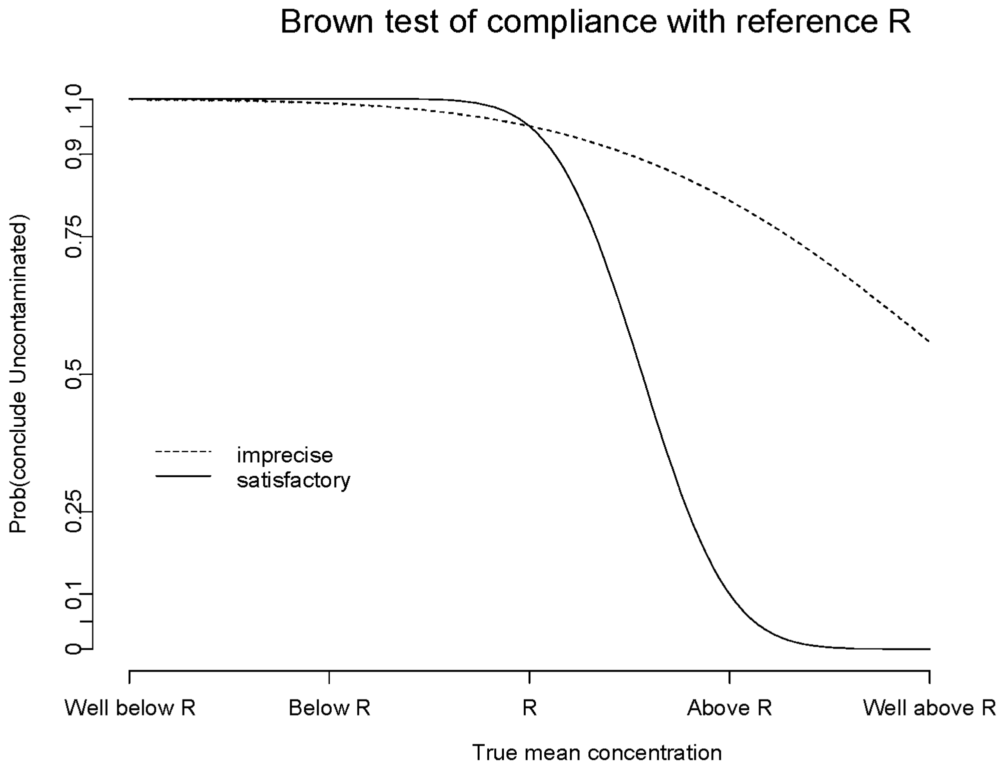

Figure 4A.1a. The probability of concluding *Uncontaminated* as the true mean concentration varies, using the brown test of compliance with reference value *R*.

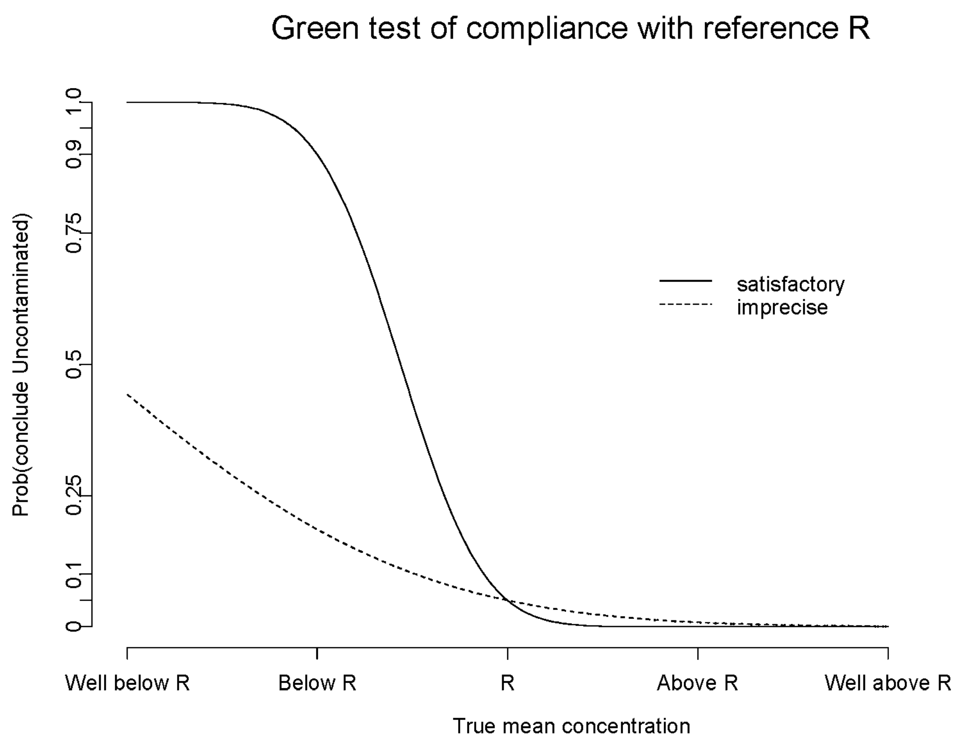

Figure 4A.1b. The probability of concluding *Uncontaminated* as the true mean concentration varies, using the green test of compliance with reference value *R*.

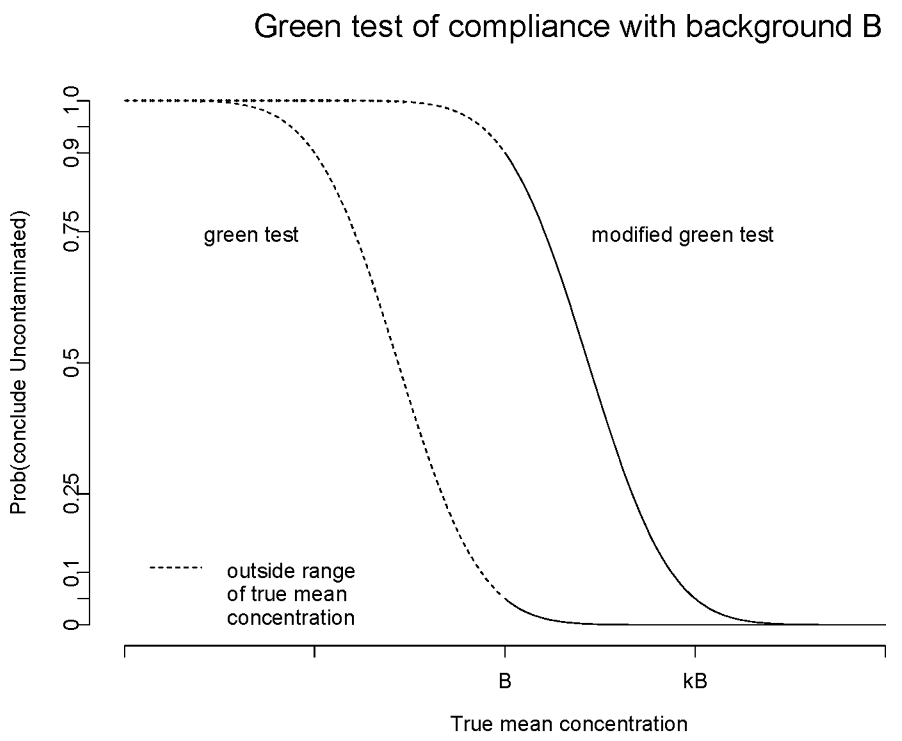

Figure A4.2a. The probability of concluding *Uncontaminated* as the true concentration varies, using the green test of compliance with background concentration *B*, and also a modified green test.

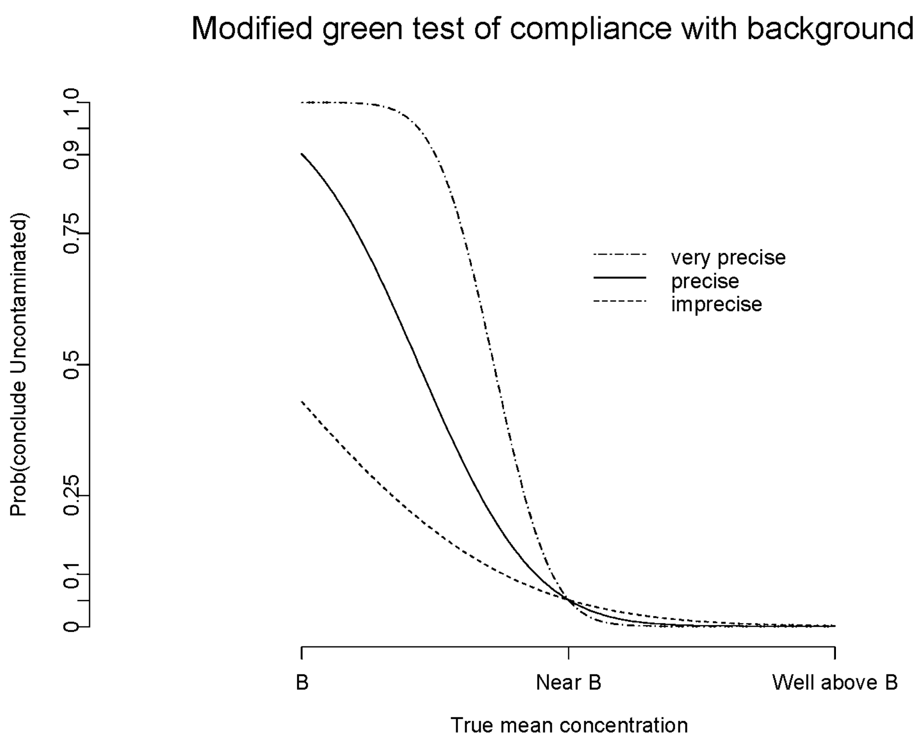

Figure 4A.2b. The properties of a modified green test of compliance with a near-background reference concentration in relation to three levels of monitoring precision.

Is it worse to conclude *Uncontaminated* when the true mean concentration is just above the reference concentration or to conclude *Contaminated* when the true mean concentration is just below the reference concentration?

This question will often be contentious. Fox (2001) recognized that neither the green test nor the brown test will usually satisfy all interested parties and proposed a hybrid test that provided some balance between the two. The modified green test provides a similar balance and is constructed by introducing a *Near-reference concentration* that is "sufficiently near to the reference concentration" in terms of its environmental impact.

Possible contexts include food and environmental standards that have some margin of safety incorporated into their construction.

3 Choice of *Near-Reference Concentration*: the Implications of the Precision of the Monitoring Programme

The choice of any *Near-reference concentration* should be based on scientific considerations. However, the choice will only be practical if we can successfully test whether the reference concentration has been achieved.

To be specific, we will discuss the construction of a *Near-background concentration*. This will be practical if there is a sufficiently high probability of concluding *Uncontaminated* when the true mean concentration is at or near background. As we have seen, this depends not only on the form of test, the choice of the *Near-background concentration*, and the value of α, but also on the precision of the monitoring programme. If the precision is too poor, then even when background concentrations have been reached, there may be only a small probability of concluding *Uncontaminated* (Figure 4A.1a).

Ideally, a monitoring programme would be designed so that its precision (i.e., standard error $se = \sigma_\mu$) is compatible with the prescribed *Near-background concentration*. However, the cost of achieving this precision might be too great. In this case, increased environmental costs might be traded against reduced monitoring costs by identifying a higher *Near-background concentration* corresponding to an achievable level of precision.

For example, consider the *Background Reference Concentration* for mercury in flatfish defined by OSPAR (1997) to be a range of 0.03–0.07 mg kg^{-1} wet weight. For demonstration, suppose the scientific advice is that an acceptable *Near-background concentration* would be twice the mid-point of this range, giving a value of 0.1 mg kg^{-1}. Then a target for σ_μ might be that with $\alpha = 5\%$, there should be a probability of 90% of concluding *Uncontaminated* when the true mean equals 0.05 mg kg^{-1}. The probability curve corresponding to this target standard error is shown by the solid line in Figure A4.3. If this level of precision cannot be achieved, Figure A4.3 shows the probability curves corresponding to increasing values of σ_μ. These are constructed with a corresponding *Near-background concentration* such that there is always a probability of 90% of concluding *Uncontaminated* when the true mean equals 0.05 mg kg^{-1}.

In a report to the Royal Commission on Environmental Pollution, Barnett and O'Hagan (1997) argued that *an ideal standard* [i.e., the *Near-background* concentration] *should not be set without also setting a standard for its statistical verification* [i.e., σ_μ]. Certainly, making the relationship between them explicit as in Figure A4.3 should help to avoid creating standards that have only a small probability of ever being met as measured by data generated from monitoring programmes.

4 Application: Mercury Concentrations in Plaice from Liverpool Bay

A monitoring programme may be concerned with more than just testing whether the *Near-background concentration* has been achieved. There may be a sequence of objectives, each becoming more, or less, relevant depending on the trend in contaminant concentrations (cf., Nicholson and Fryer, 1997). For example, during the early years of a monitoring programme, the emphasis may be on evidence of a decline, or for a reduction relative to some upper reference value. After a period of recovery, however, interest may switch to testing for compliance with a more stringent target such as a background concentration. The green and modified green tests described here lend themselves to this cascading sequence of tests, with a new test initiated as each objective is achieved.

This is demonstrated using data for mercury concentrations in the muscle of commercial-sized plaice (*Pleuronectes platessa*) from Liverpool Bay in the Irish Sea. These data and their background are discussed and analysed in detail in Nicholson and Fryer (2002). We consider both a background test and a test of an upper reference value.

We use the *Background Reference Concentration* for mercury in flatfish of 0.03–0.07 mg kg^{-1} wet weight described in Section 3. We will adopt a modified green test of compliance with near-background, and again define the corresponding *Near-background concentration* to be twice the mid-point of this range, giving a value of 0.1 mg kg^{-1}.

For an upper reference, we adopt a value of 0.3 mg kg^{-1}. This value was set by the European and Paris Commissions (EC Directive No. 82/176) as a quality standard for a representative sample of locally caught fish, and has since been used by OSPAR as a guide to an *Upper* level for mercury (CEFAS, 2000). We will adopt a simple green test for compliance for this reference point.

There are now three possible decisions we could make, which we label as *Uncontaminated*, *Contaminated*, and *Highly Contaminated*, related to the corresponding tests as follows:

Test outcome	Decision	Probable state of environment
$UCL_T > 0.3$	Highly Contaminated	$\mu_T > 0.3$
$0.1 < UCL_T \leq 0.3$	Contaminated	$0.1 < \mu_T \leq 0.3$
$UCL_T \leq 0.1$	Uncontaminated	$\mu_T \leq 0.1$

To estimate μ_T and derive UCL_T, we use the method described by Fryer and Nicholson (1999). For a series of annual indices $y_1, y_2, \ldots, y_T$, we assume that μ_t, the mean in year t, can be represented by a smooth function $f(t)$, and that this can be estimated using a LOESS smoother. The value predicted by the smoother in year T gives $\hat{\mu}_T$, which is combined with the residual variance to give the UCL_T. The results are shown in Figure A4.4. The upper 95% confidence limit for the final year (1995) is 0.163 mg kg^{-1}, so we reject H_0 (*Highly Contaminated*) and accept H_1 (*Contaminated*), but we do not accept H_2 (*Uncontaminated*).

5 Discussion

We have described a modified green test, a very simple, precautionary method for assessing whether contaminant concentrations comply with some reference concentration. In particular, we have focused on testing whether the OSPAR policy objective of achieving concentrations near background has been met. It relies on defining a *Near-background concentration*, below which concentrations are deemed acceptably low.

As demonstrated using a time series of mercury concentrations in plaice from Liverpool Bay, the method can easily be incorporated into a general trend assessment method based on smoothers.

Choosing the *Near-background* concentration will always be difficult. Scientific and policy issues will obviously be important, but the precision of the monitoring programme must also be taken into account. Iterating between what we want to detect and what we can detect provides a practical scheme for identifying a workable *Near-background concentration*. Further, linking the *Near-background concentration* to a specific performance target for monitoring precision provides direct pressure to maintain monitoring quality. With poor monitoring, it becomes increasingly difficult to conclude that results are *Uncontaminated*, even when they are.

The choice of an upper reference value might be less complicated, although there may be several candidates. Our example uses a value taken from a specific historical context, but alternatives include a historic mean coinciding with some management intervention, or the *Ecotoxicological Assessment Criteria* introduced by OSPAR (2000). However, the simple green test used here for the upper reference may be overly protective if some environmental protection has already been incorporated in the construction of the upper reference. In this case, the less-precautionary modified green test against an appropriate *Near-upper reference concentration* should be considered.

Acknowledgement

This report was prepared by M. Nicholson, CEFAS Lowestoft Laboratory, Lowestoft, UK, and R. Fryer, FRS Marine Laboratory, Aberdeen, UK.

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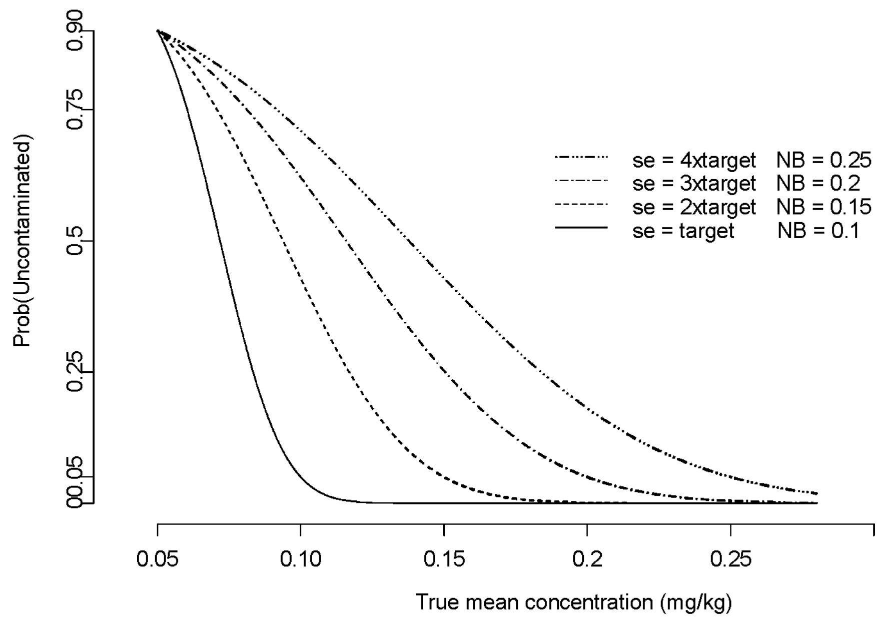

Figure A4.3. Probability curves corresponding to the target standard error (solid line) and to increasing values of standard error.

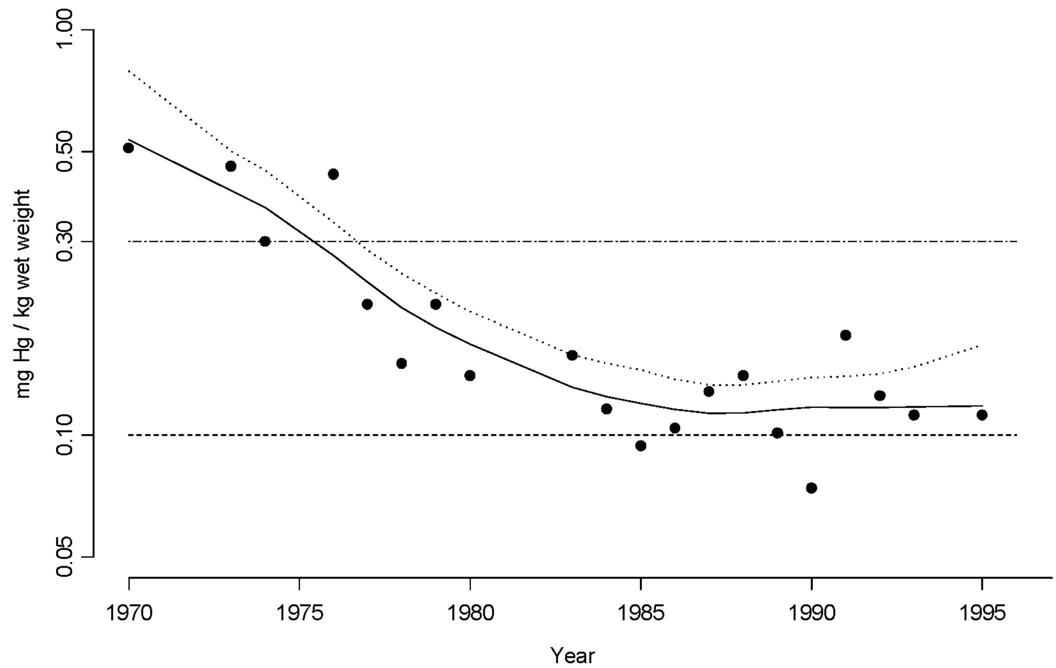

Figure A4.4. Application of the green test to data on mercury concentrations in plaice from Liverpool Bay, using a Near-background concentration of 0.1 mg kg^{-1} and an upper reference of 0.3 mg kg^{-1}.

ANNEX 5: TEMPORAL TREND ANALYSIS: STATISTICAL METHODS FOR IDENTIFYING A BREAK IN THE TREND

In this annex, a simple method for the identification of change-points in the trend is discussed. Three models are considered:

1) Linear trend;
2) Linear trend with seasonality;
3) LOESS trend.

1 Identification of Changes in the Linear Trend

The model applied is:

$$y_t = a_{s(t)} + b_{s(t)} p_t + error$$

where

$$s(t) = \begin{cases} 1 & p_t \le b \\ 2 & p_t > b \end{cases},$$

$t = 1,\ldots,n$

and b denotes the time of the change-point in the trend. The error components are assumed to be stochastically independent and identically Normally distributed.

If the trend line is continuous, the continuity condition holds:

$$a_1 + b_1 b = a_2 + b_2 b .$$

In matrix notation, the model will be re-parameterized and extended to non-equidistant time periods:

$$Y = X_b \beta + \varepsilon$$

where $Y = (Y_1,\ldots,Y_n)^T$,

$$X_b = \begin{pmatrix} 1 & p_1 - b & 0 \\ \vdots & \vdots & \vdots \\ 1 & p_k - b & 0 \\ 1 & 0 & p_{k+1} - b \\ \vdots & \vdots & \vdots \\ 1 & 0 & p_n - b \end{pmatrix},$$

$$p_k \le b \le p_{k+1} ,$$

$4 \le k \le n-3$ and

$\beta = (\beta_1,\beta_2,\beta_3)^T$.

Algorithm

The proposed algorithm for the identification of a break in the trend is as follows:

Step A: Selection of time intervals

Select b out of $\{p_4,\ldots, p_{n-3}\}$ so that the sum of squared residuals

$$SS_{2sections} = \left(Y - X_b\left(X_b^T X_b\right)^{-1} X_b^T Y\right)^T \left(Y - X_b\left(X_b^T X_b\right)^{-1} X_b^T Y\right)$$

attains its minimum.

Step B: Examination of the significance of the break

Calculate the sum of squared residuals

$$SS_{lin} = \left(Y - X\left(X^T X\right)^{-1} X^T Y\right)^T \left(Y - X\left(X^T X\right)^{-1} X^T Y\right)$$

for the simple linear regression model

$$X = \begin{pmatrix} 1 & p_1 \\ 1 & p_2 \\ \vdots & \vdots \\ 1 & p_n \end{pmatrix}$$

and the test statistic

$$F = \frac{\left(SS_{lin} - SS_{2sections}\right)/ f_1}{SS_{2sections} / f_2}$$

Under the null hypothesis that there is no break in the trend, the test statistic F can be considered approximately F-distributed with $(1,n-3)$ degrees of freedom, if b is fixed. However, b has to be estimated, thus reducing the actual degrees of freedom of $SS_{2sections}$ and increasing the actual significance level. In order to correct for this exceedance of the significance level, it is proposed to use an F distribution with $f_1 = 2$ and $f_2 = n-4$ degrees of freedom. This correction provides sensible actual significance levels for $n = 10,...,50$; see Table A5.1.

Table A5.1. Actual significance level (for a formal significance level of 5%, calculated with a simulation study with 3,000 runs).

Sample size	degrees of freedom (f_1,f_2)	
n	$(1,n-3)$	$(2,n-4)$
7	5%	0.7%
8	7.3%	1.5%
10	10.5%	2.6%
12	12.6%	3.5%
15	14.4%	4.3%
20	16.8%	5.3%
30	17.1%	5.5%
40	19.0%	6.4%
50	21.9%	7.1%
100	23.4%	8.6%

Hence, if F exceeds the 95% quantile of the F distribution with $(2,n-4)$ degrees of freedom, it can be concluded that there is a break in the trend.

Example

An example of a significant change-point is presented in Figure A5.1.

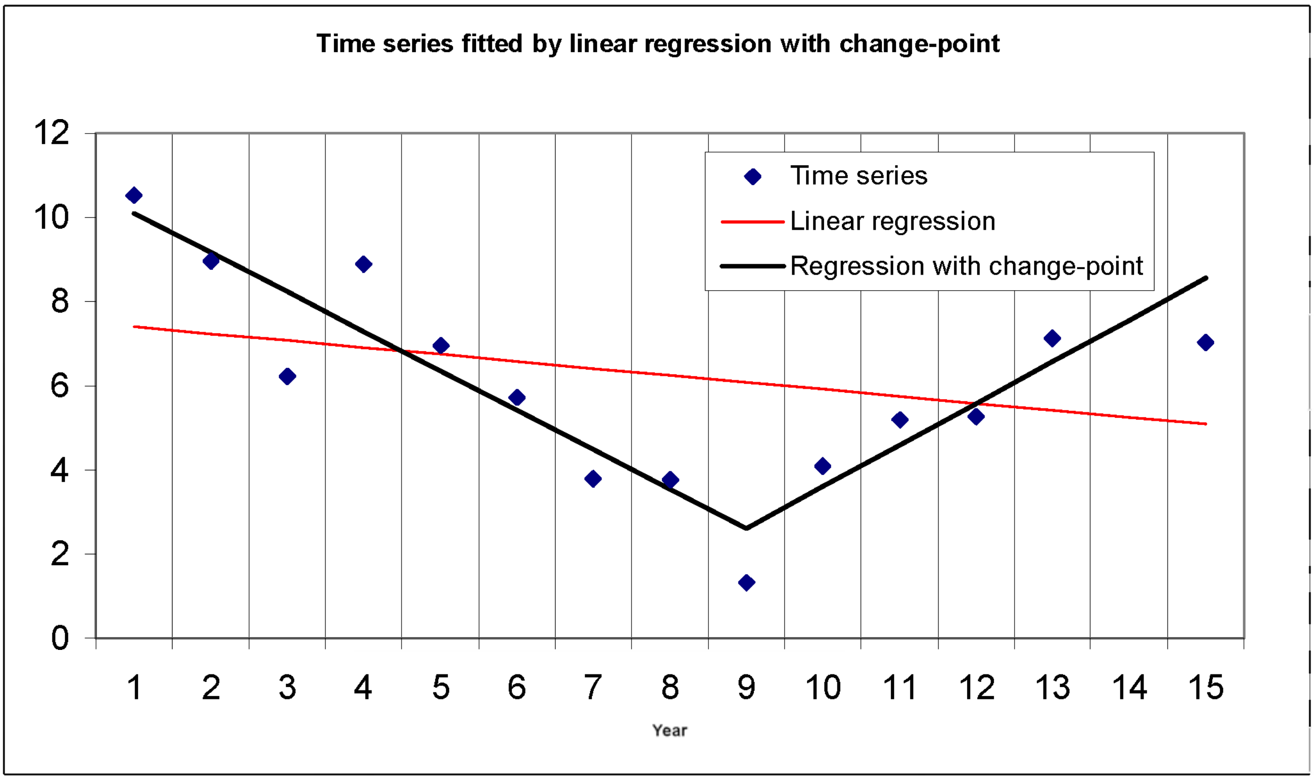

Figure A5.1. An example of a significant change-point.

2 Identification of Changes in the Linear Trend with Seasonality

For half-yearly and quarterly data, the examination of a trend reversal with seasonality can be obtained by extending the design matrix X_b and the design matrix for a linear trend with seasonal dummies s_t:

$$X_b = \begin{pmatrix} 1 & p_1 - b & 0 & s_1 \\ \vdots & \vdots & \vdots & \vdots \\ 1 & p_k - b & 0 & s_k \\ 1 & 0 & p_{k+1} - b & s_{k+1} \\ \vdots & \vdots & \vdots & \vdots \\ 1 & 0 & p_n - b & s_n \end{pmatrix} \text{ and}$$

$$X = \begin{pmatrix} 1 & p_1 & s_1 \\ 1 & p_2 & s_2 \\ \vdots & \vdots & \vdots \\ 1 & p_n & s_n \end{pmatrix}$$

where in case of half-year data

$$s_t = \begin{cases} 0 & t \text{ represents winter period} \\ 1 & t \text{ represents summer period} \end{cases}$$

and in case of quarterly data

$$s_t = \begin{cases} (0,0,0) & t \text{ represents winter} \\ (1,0,0) & t \text{ represents spring} \\ (0,1,0) & t \text{ represents summer} \\ (0,0,1) & t \text{ represents autumn} \end{cases}.$$

The determination of the time intervals is as described in the preceding sub-section.

The test statistic can be calculated

$$F = \frac{(SS_{lin} - SS_{2sections})/2}{SS_{2sections} / (n-m)}.$$

where $m = 5$ for half-yearly data and

$m = 7$ for quarterly data.

Under the null hypothesis that there is no break in the trend, the test statistic F can be considered approximately F-distributed with $(2,n-m)$ degrees of freedom. Hence, if F exceeds the 95% quantile of the F distribution with $(2,n-m)$ degrees of freedom, it can be concluded that there is a break in the trend.

Example

An example of a significant change-point for quarterly data is presented in Figure A5.2. It appears that there is only a small effect of the change-point. However, the sum of squared residuals is reduced by almost 50%, thus indicating that the change-point is highly significant.

3 Identification of Changes in the LOESS Trend

The method can simply be extended using the LOESS approach: Use matrix X_b instead of the simple linear regression design matrix and perform weighted regression with weights specified for each point separately:

The smoother matrix S of the LOESS smoother is constructed from a series of weighted regressions constructed for each year p_t using any data that fall within a range $p_t \pm \Delta_t$. The weight of the observation y_i in the local regression for year p_t is determined by the distance between year p_i and year p_t,

$$w_{ti} = \begin{cases} \left(1 - \left(\frac{|p_t - p_i|}{\Delta_t}\right)^3\right)^3 & \text{for } 0 \le \frac{|p_t - p_i|}{\Delta_t} \le 1 \\ 0 & \text{otherwise} \end{cases}.$$

The width of Δ_t controls the amount of smoothing, and Nicholson and Fryer (1999) choose Δ_t to include data from a fixed span of years. For change-point models, it should be guaranteed additionally that all of these data are either before or after the change-point, i.e.,

$$\Delta_t = \max\left\{\frac{\text{span}+1}{2}, \text{span} - \min\{p_t - p_1, p_n - p_t, |p_t - b|\}\right\}$$

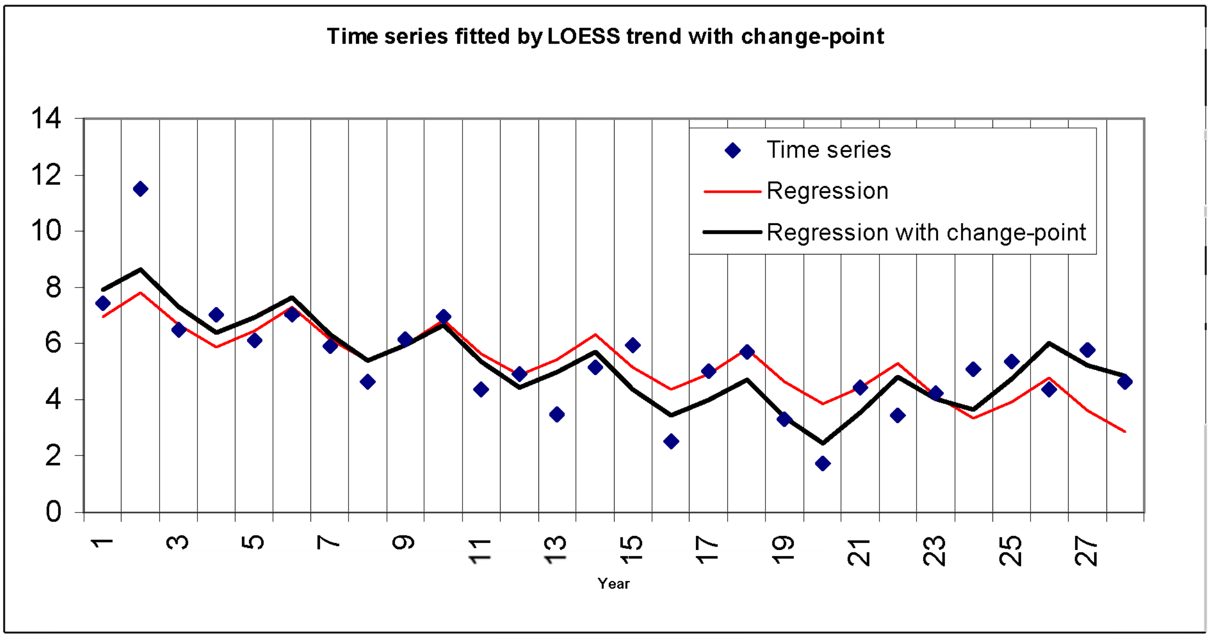

Figure A5.2. An example of a significant change-point for quarterly data.

In this paper, a span of eleven years is used. Then, writing W_t for the corresponding diagonal matrix of weights,

$$W_t = \begin{pmatrix} w_{t1} & 0 & \cdots & 0 \\ 0 & w_{t2} & \ddots & \vdots \\ \vdots & \ddots & \ddots & 0 \\ 0 & \cdots & 0 & w_{tn} \end{pmatrix},$$

the value of the smoother in year p_t is given by

$$S_t'Y,$$

where S_t' is the tth row of

$$X_b\left(X_b'W_tX_b\right)^{-1}X_b'W_t \quad \text{if none of the}$$

columns of the matrix W_tX_b is null, or

$$X\left(X'W_tX\right)^{-1}X'W_t \quad \text{else,}$$

where X denotes the classical design matrix for ordinary regression.

This allows an extended ANOVA, comparing the fits of

- linear trend
- LOESS trend
- linear trend with break
- LOESS trend with break.

Example

For the example presented in Figure A5.3, the following sum of squared residuals were calculated. The degrees of freedom were calculated according to the procedure described in Fryer and Nicholson (1999). For the LOESS trend, however, the degrees of freedom were obtained by adding two degrees of freedom for the LOESS trend with change-point.

Model	**df of residuals**	**Sum of squared residuals**
Linear trend	25	31.4
LOESS trend	>22.8	5.32
Linear trend with change-point	24-1	20.3
LOESS trend with change-point	21.8-1	3.69

Using these sums of squares, it turns out that both the change-point and the non-linear trend component in the LOESS trend are significant.

Acknowledgement

This paper was prepared by S. Uhlig, quodata, Dresden, Germany.

Reference

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(6): 779–790.

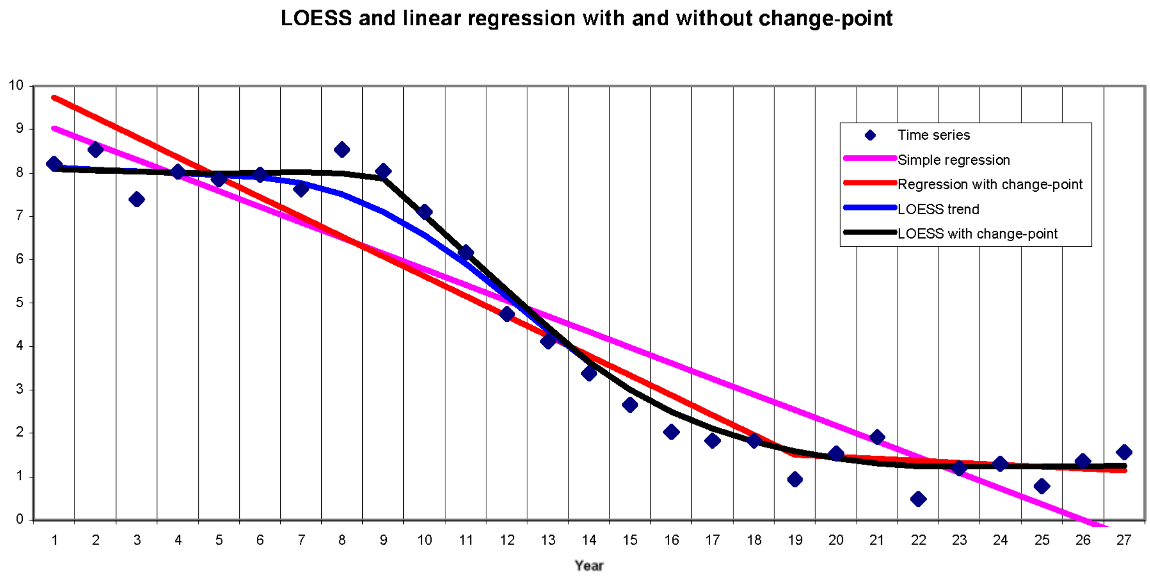

Figure A5.3. LOESS and linear regression with and without change-point.

ANNEX 6: REVIEW NOTE ON PHENYLUREA HERBICIDES IN THE MARINE ENVIRONMENT

Abstract

The occurrence of phenylurea herbicides in the North Sea and Baltic Sea is described. In addition, the concentrations observed in the river Elbe, the major input source for the German Bight, are presented. It is shown that especially the compounds Diuron and Isoproturon can be detected in the whole area under investigation. As a generally applicable analytical procedure for the determination of phenylurea herbicides in water, a method is described that is based on a solid phase extraction (SPE) and a HPLC-MS-MS quantification method. This procedure is sensitive and selective enough to determine the low concentrations found in the open sea (down to 0.1 ng l^{-1}).

1 Introduction

Phenylurea herbicides are an important class of modern herbicides that are produced and used in Europe in large quantities. Due to the large quantities used and the fact that some of these compounds exhibit quite a slow degradation rate in the environment, they deserve special attention in environmental monitoring. Diuron and Isoproturon are therefore on lists of possible priority pollutants (EU, OSPAR). Because of their good water solubility and their persistence, they can reach the marine environment.

In this review, the following compounds are dealt with: Diuron, Isoproturon, Chlortoluron, Fenuron, Linuron, and Monolinuron. The chemical structures are depicted in Figure A6.1; some relevant properties are listed in Table A6.1. Table A6.2 summarizes some production and application amounts.

Owing to their herbicidal properties, the phenylureas exhibit high toxicities, especially against algae and water plants. Therefore, in Table A6.3 these toxicities are listed. Diuron and Isoproturon were shown to be toxic for fishes and small crustaceans also.

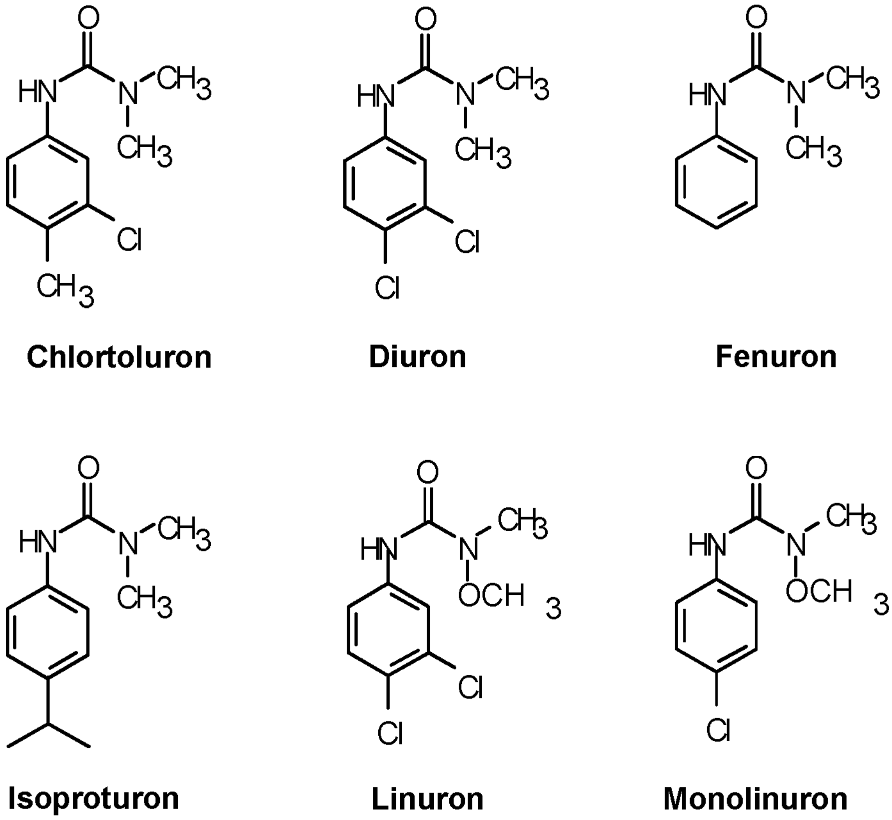

Figure A6.1. Chemical structures of the phenylurea herbicides under investigation.

Table A6.1. Properties of phenylurea herbicides.

Compound	Log P_{OW}	Degradation in water (half-life)	Degradation in soil (half-life)
Chlortoluron	2.41		
Diuron	2.82	stable	3 to 5 m
Fenuron	0.96		
Isoproturon	2.5	stable	6–23 d
Linuron	2.76		
Monolinuron	2.3		

Table A6.2. A summary of production and application amounts.

Compound	Production in EU (t yr^{-1})	Use in Germany (t yr^{-1})
Chlortoluron		
Diuron	3,000–15,000	200–500
Fenuron		
Isoproturon	10,000–50,000	> 1000
Linuron		
Monolinuron		

Table A6.3. Toxicity data on phenylurea herbicides.

Compound	NOEC Algae (µg l^{-1})	LC50 fish	EC50 invertebrate
Chlortoluron		30–50 mg l^{-1}	
Diuron	0.46	5–25 mg l^{-1}	1–12 mg l^{-1}
Fenuron			
Isoproturon	2	30–100 mg l^{-1}	507 mg l^{-1}
Linuron		3	120 µg l^{-1}
Monolinuron		56–74 mg l^{-1}	32 mg l^{-1}

2 Analysis of Phenylurea Herbicides

Because of their good water solubility and their polarity (low log P_{OW} values < 3), the water phase is the preferred monitoring matrix. Extraction from water samples is preferably done by solid phase extraction (SPE). Not all phenylurea herbicides can be analysed by GC-MS. Therefore, HPLC is used as a more general analysis technique. As UV-detection is not very sensitive or selective, MS or MS-MS detection is the preferred technique.

Water samples are sampled in 10 l glass bowls. The water is acidified with 10 ml of hydrochloric acid (25%) to a pH of 2 to 3, and a solution of deuterated internal standards (d_6-Diuron) is added. 6 l of this sample are pumped in batches of 2 l over three separate cartridges, each filled with 600 mg of Chromabond HR-P resin (Macherey-Nagel, Düren, Germany). After washing with pure, deionized water and drying in a stream of nitrogen, the cartridges are eluted with 10 ml dichloromethane and 10 ml acetone each.

After concentrating the combined extracts to 75 µl, the sample solution is mixed with 50 µl of water. 20 µl of this solution is injected into the HPLC-MS-MS system (Agilent 1100 Series autosampler, binary pump, column oven 23 °C, and Applied Biosystems API 2000 LC-MS-MS system, ESI source). A phenyl phase column (Synergi Polar-RP, Phenomenex) is used (2 mm, 7.5 cm) with a flow rate of 200 to 250 µl min^{-1}. Eluent A consists of water with 10 mM ammonium acetate and 10 mM acetic acid, pH 4.5; eluent B is methanol with the same concentrations of buffering compounds. The following gradient is used:

0 min, 80% A, 200 µl min^{-1}
12 min, 50% A, 250 µl min^{-1}
25 min, 30% A, 250 µl min^{-1}
36 min, 5% A, 250 µl min^{-1}
45 min, 5% A, 250 µl min^{-1}

The mass spectrometric parameters, the overall recoveries, and limits of determination are summarized in Table A6.4.

The procedure was validated and controlled by internal and external QA procedures. Internal QA covers determination of procedural blanks (smaller than LOD), recoveries by standard addition (75% to 90%), tests for calibration linearity and determination of the detection limit (s/n 10). External QA was proven by participation in two QUASIMEME intercomparison exercises; all Z scores were < 2.

2 Results

The data presented here were obtained within the German monitoring programme and cover the years 2000 to 2002.

Table A6.5 summarizes the overall data, covering data from the river Elbe, the German Bight, North Sea, and Baltic Sea. As expected, the maximum values are observed in the river Elbe. The two most important compounds are Diuron and Isoproturon, followed by Chlortoluron, Linuron, and Fenuron. Monolinuron is found in the Elbe only.

The spatial distribution of the different compounds is quite similar and in coastal areas parallels the freshwater content of the sea water. As Figure A6.2 indicates, there are differences concerning the steepness of the gradient, possibly indicating differences in the degradation. Chlortoluron, Diuron, and Isoproturon seem to be more stable because of the less steep gradient. However, the data are not adequate for a clear statement.

Table A6.4. Analytical parameters.

Compound	MS-MS Transitions	Overall recovery %	Limit of determination (ng l^{-1})
Chlortoluron	212.9 / 72.1	80	0.35
Diuron	232.9 / 72.1	90	0.10
Fenuron	165.0 / 72.0	75	0.06
Isoproturon	207.0 / 72.1	90	0.04
Linuron	249.0 / 160.2	90	0.14
Monolinuron	215.0 / 126.1	90	0.08

Table A6.5. Overview of the concentrations of phenylurea herbicides observed (ng l^{-1}).

	Area	Count	Average	Median	Minimum	Maximum
Chlortoluron						
	Elbe	17	4.20	3.85	2.03	7.58
	North Sea	52	1.32	1.34	< 0,35	3.20
	Baltic Sea	8	0.30	0.22	0.15	0.77
Diuron						
	Elbe	25	22.19	13.56	7.74	124.16
	North Sea	89	3.63	3.70	0.10	14.32
	Baltic Sea	18	3.04	0.97	0.42	13.34
Fenuron						
	Elbe	16	2.88	2.62	1.39	5.39
	North Sea	28	0.18	0.17	<0,06	0.46
	Baltic Sea	1	0.39	0.39	0.39	0.39
Isoproturon						
	Elbe	25	11.46	6.92	2.67	36.90
	North Sea	86	2.33	2.14	0.05	8.28
	Baltic Sea	18	0.61	0.43	0.20	2.49
Linuron						
	Elbe	19	1.44	1.07	0.13	3.67
	North Sea	7	0.23	0.21	<0,14	0.54
Monolinuron						
	Elbe	5	0.36	0.30	0.13	0.87
	North Sea	5	0.19	0.21	<0,08	0.41

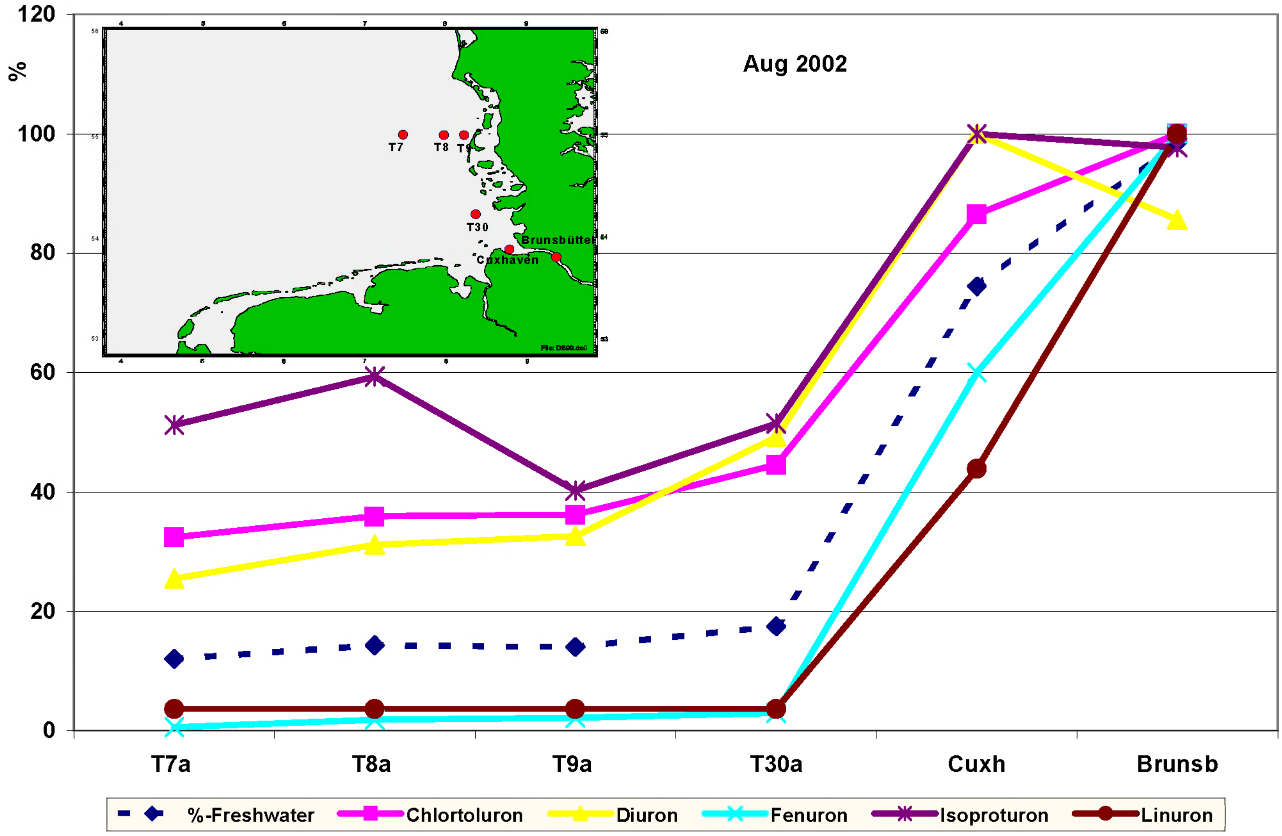

Figure A6.2. Spatial concentration gradients of herbicides in the Elbe plume (% of the maximum concentration in the Elbe).

The geographical distributions of Diuron and Isoproturon in the North Sea are shown respectively in Figures A6.3 and A6.4. Both compounds are found in relatively high concentrations in coastal areas, especially at the estuaries of the large rivers Elbe and Rhine (Diuron 1–10 ng l^{-1}, Isoproturon 1–5 ng l^{-1}). In the central North Sea or the Atlantic (Bay of Biscay), concentrations are very low (Isoproturon 0.05 ng l^{-1}, Diuron 0.2 ng l^{-1}).

In deep-water (500 m) samples of the Atlantic and Skagerrak, significantly lower values are observed compared to surface-water samples.

Diuron has been analysed in the marine environment during the past five years in several studies in connection with its use as a booster biocide in antifouling paints for small ships (Comber *et al.*, 2002; Martinez and Barcelo, 2001; Gimeno *et al.*, 2001; Piedra *et al.*, 2000; Martinez *et al.*, 2000; Voulvoulis *et al.*, 1999a, 1999b; Ferrer and Barcelo, 1999; Thomas, 1998). Most studies were performed in marinas on the UK and Spanish coasts, where concentrations ranged from 3–2000 ng l^{-1}. In summer, generally higher values are observed than in winter, due to a higher ship density. In the Ebro delta, concentrations of < 10–117 ng l^{-1} are reported (Ferrer and Barcelo, 1999). Concentrations in the open sea have not been determined as yet. In the Baltic Sea (Figures A6.5 and A6.6), concentrations are in a similar range as in the North Sea. Concentrations in the Baltic do not reach the low values observed in the open North Sea or Atlantic; this can be explained by the more enclosed sea area of the Baltic Sea. In coastal areas, there seem to be regional differences between Isoproturon and Diuron.

While Isoproturon shows the highest values (up to 2.5 ng l^{-1}) in the mouth of the river Odra, indicating this as an input source, Diuron is found in the highest concentrations in the western part of the Baltic, in the Kiel and Flensburg bights. Whether these elevated concentrations are caused from its possible use as an antifouling agent on pleasure boats can be speculated but was not proved by the measurements.

The temporal development in the German Bight over the years 2000 to 2002 for Diuron and Isoproturon is shown in Figures A6.7 and A6.8. The values show quite a high variability and the time series are too short to observe any statistically relevant trend. The same is true for seasonal fluctuations within a year.

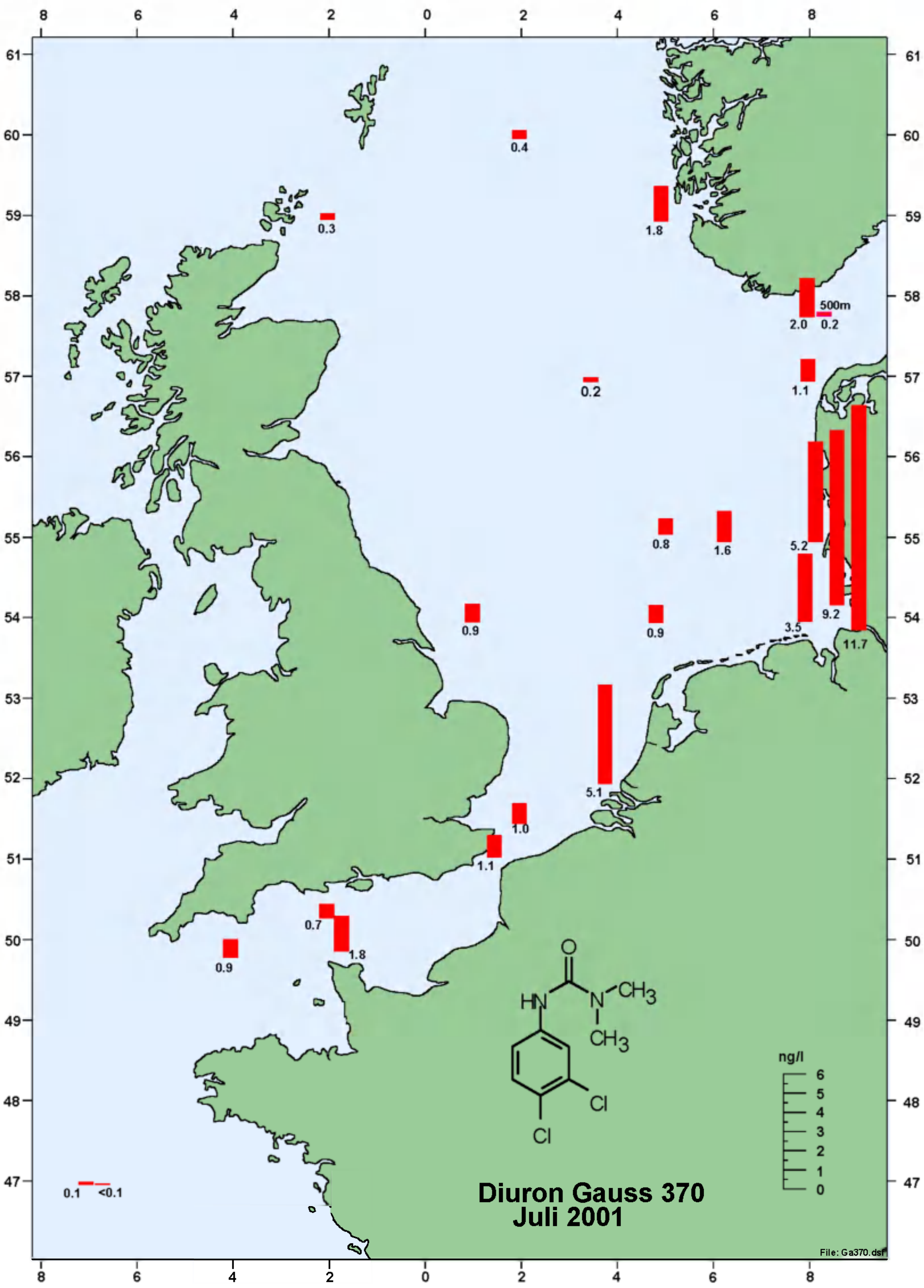

Figure A6.3. Concentrations of Diuron (ng l^{-1}) in the North Sea in July 2001 (unless otherwise stated, surface water (5 m)).

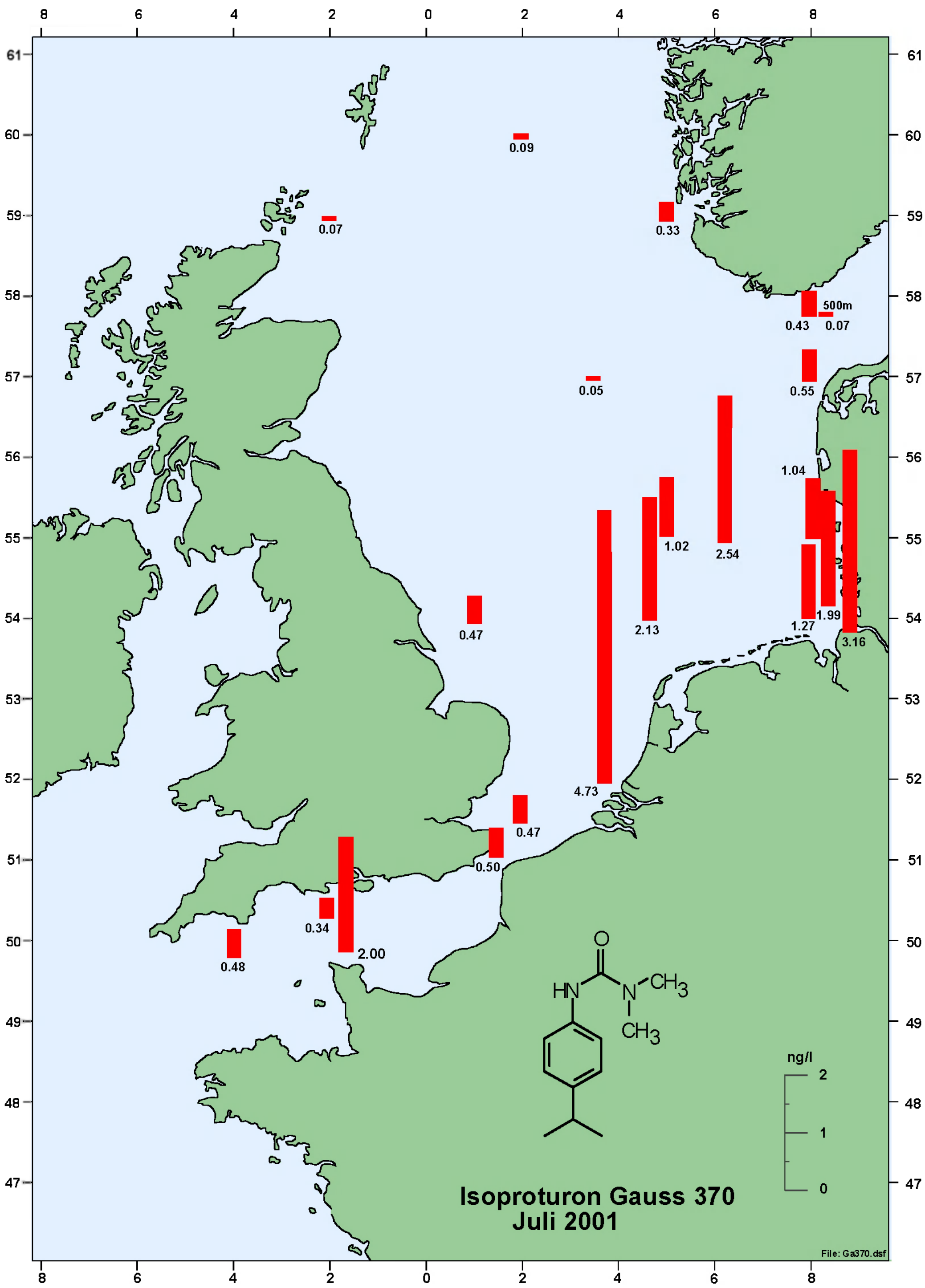

Figure A6.4. Concentrations of Isoproturon (ng l^{-1}) in the North Sea in July 2001 (unless otherwise stated, surface water (5 m)).

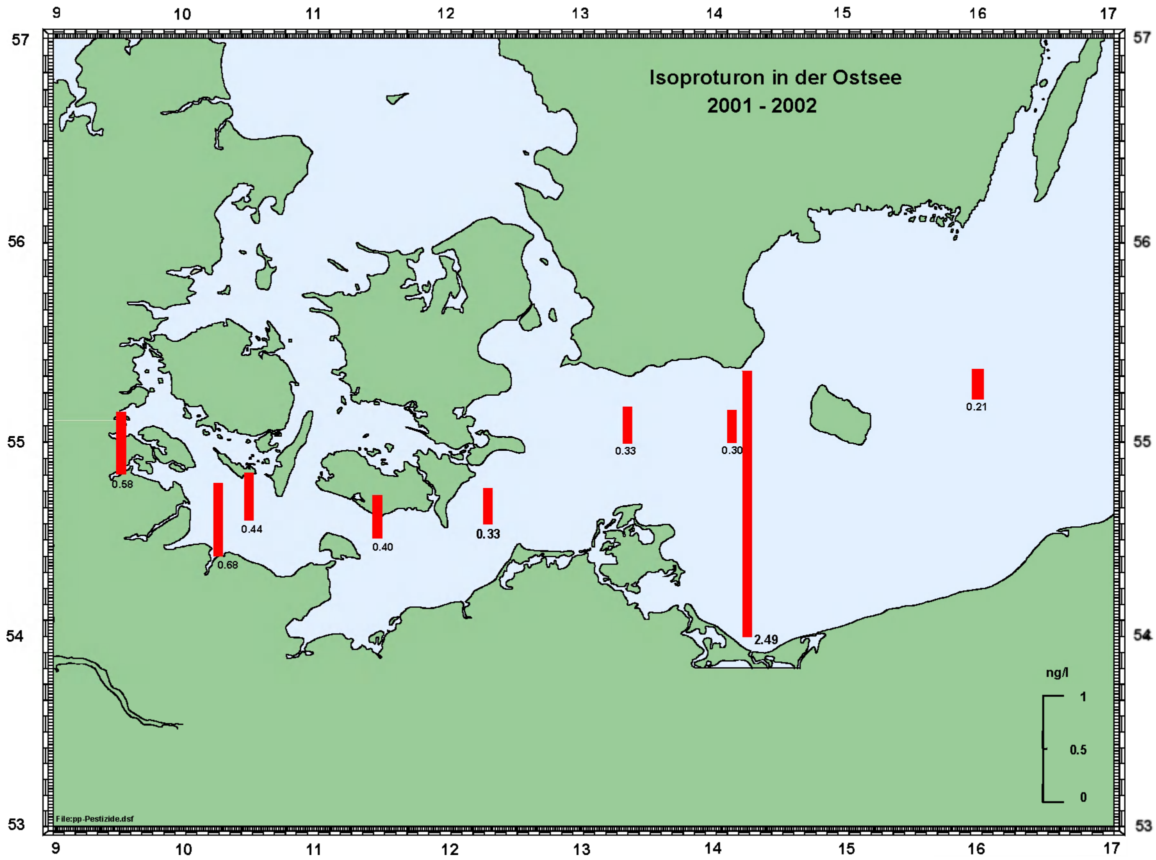

Figure A6.5. Concentrations of Isoproturon (ng l^{-1}) in the Baltic Sea—average values of August 2001 and 2002 (5 m).

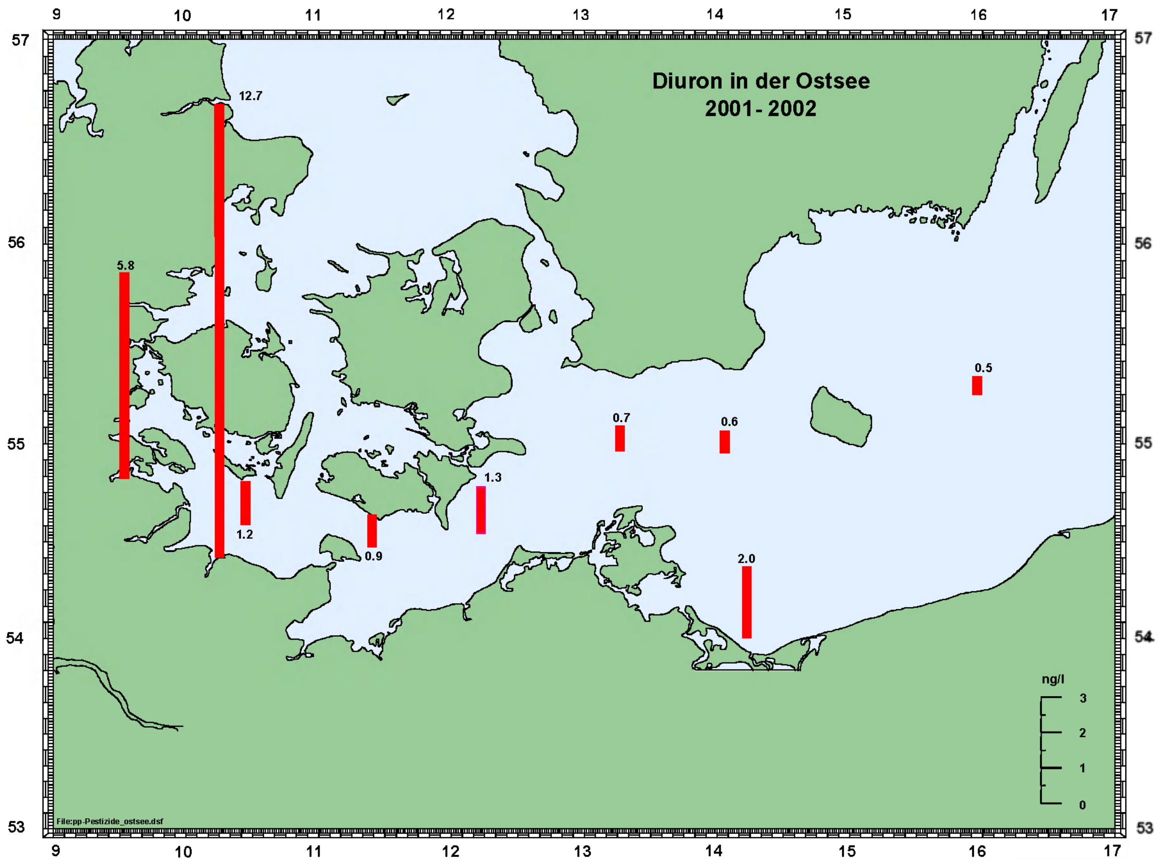

Figure A6.6. Concentrations of Diuron (ng l^{-1}) in the Baltic Sea—average values of August 2001 and 2002 (5 m).

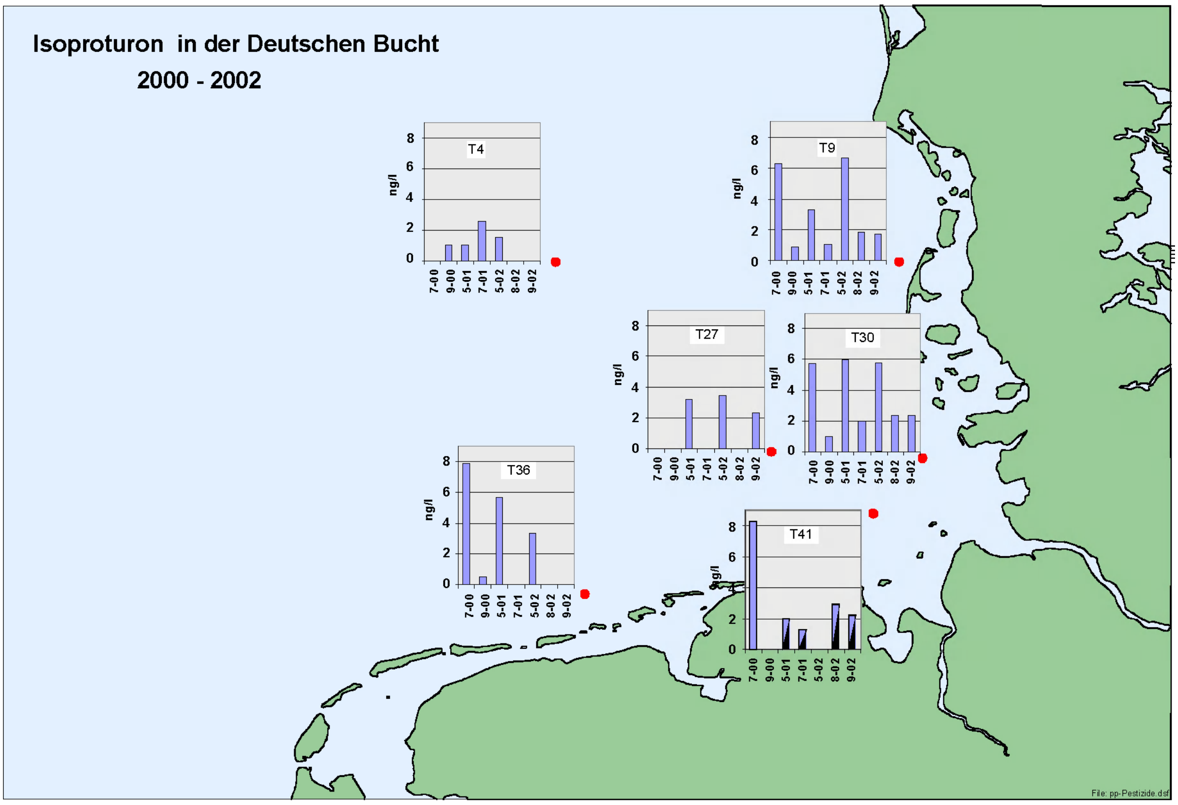

Figure A6.7. Concentrations of Isoproturon in the German Bight, 2000 to 2002 (surface water (5 m)).

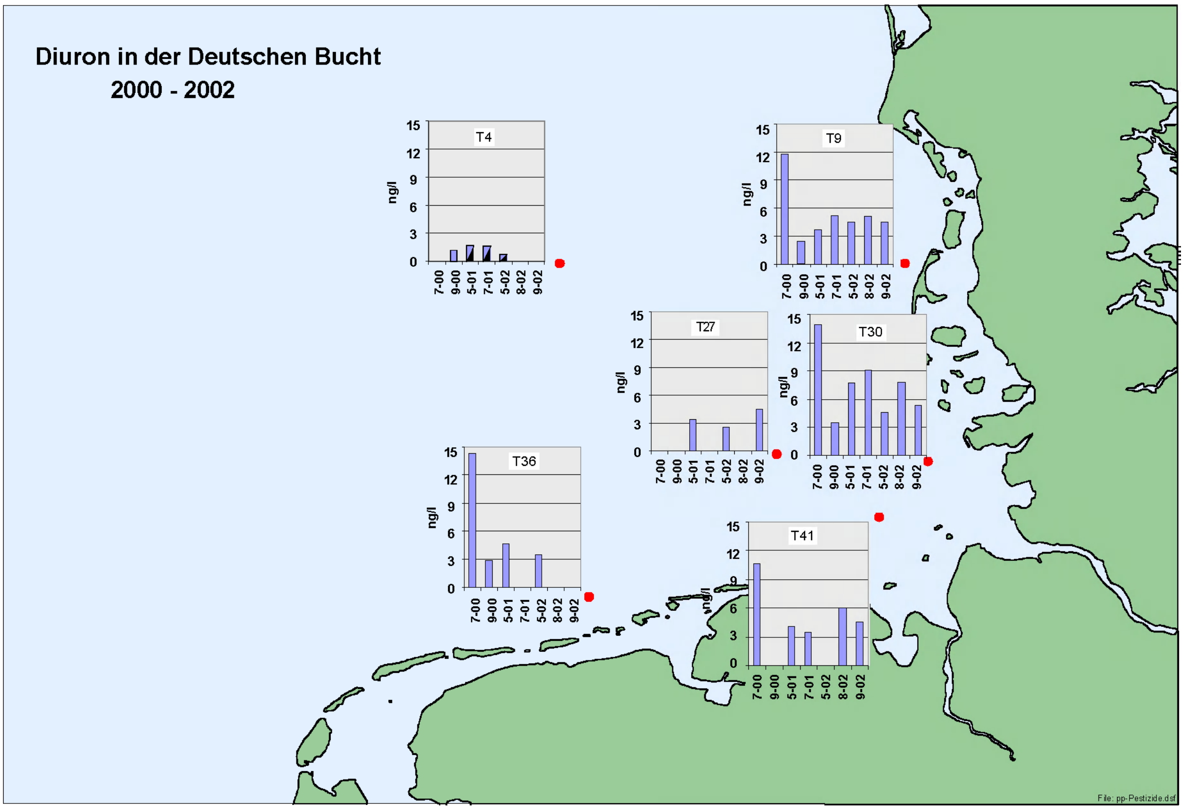

Figure A6.8. Concentrations of Diuron in the German Bight, 2000 to 2002 (surface water (5 m)).

The phenylurea herbicides have been determined in many land-based monitoring programmes and positive findings have been reported both for surface water and groundwater. Unfortunately, in many reports only information on positive findings (< LOD), samples above a limit of 0.1 µg l^{-1} and maximum values are reported.

In the river Rhine, Chlortoluron, Linuron, Diuron, and Isoproturon have been monitored; in 2000 they all showed average values of less than 50 ng l^{-1} (LOD). For Diuron, a maximum of 70 ng l^{-1}, and for Chlortoluron and Isoproturon maxima of 120 ng l^{-1} were observed (RIWA, 2002).

4 Conclusion

Several phenylurea herbicides have been detected in the marine environment. Especially Diuron and Isoproturon are found in sea water of the North Sea and Baltic Sea at concentrations which are higher than those of "classical" contaminants such as HCH isomers. Compared to lipophilic contaminants such as PCBs or PAHs, they even show 10 to 100 times higher values. As Diuron and Isoproturon are often found in surface water and are considered priority pollutants in the EU Water Framework Directive, they should be determined in future marine monitoring surveys. For Diuron there is, in addition to land-based sources, a "marine" source possibly because of its use as an antifouling paint constituent.

Acknowledgement

This report was prepared by N. Theobald, Federal Maritime and Hydrographic Agency, Hamburg, Germany.

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ANNEX 7: ICES CODE OF PRACTICE ON THE INTRODUCTIONS AND TRANSFERS OF MARINE ORGANISMS 2003

PREAMBLE

Global interest in marine aquaculture (mariculture) began to increase dramatically in the 1950s and 1960s. A natural complement to this interest was the search for fish, shellfish (molluscan, crustacean, and echinoderms), and plant species whose biology was well known and which already had achieved or could achieve success in extensive cultivation or which could be of interest in research. Once identified, these species were thus potential candidates for movement to new locations in the world for the purpose of establishing new fisheries and new mariculture resources. Such animals and plants that are not native to these new locations are defined as species transported intentionally or accidentally by a human-mediator vector into aquatic habitats outside their native range, including secondary introductions by human-mediated or natural vectors. Other terms used for such introductions are alien, exotic, invasive, foreign, non-native, immigrant, neobiota, naturalized, or non-indigenous.

While the Code of Practice was originally developed for marine aquaculture activities, in recent years, by far the largest number of introductions have been for re-stocking or enhancement purposes but the same principles should apply.

While great successes have been achieved by these activities, leading to the creation of new and important fishery and mariculture resources, three challenges have surfaced over the past several decades relative to the global translocation of species to new regions.

The first challenge lies in the ecological and environmental impacts of introduced and transferred species, especially those that may escape the confines of cultivation and become established in the receiving environment. These new populations can have an impact on native species.

The second challenge stems from the potential genetic impact of introduced and transferred species, relative to the mixing of farmed and wild stocks as well as to the release of genetically modified organisms.

The third challenge is posed by the inadvertent coincident movement of harmful organisms associated with the target (host) species. The mass transfer of large numbers of animals and plants without inspection, quarantine, or other management procedures has inevitably led to the simultaneous introduction of pathogenic or parasitic agents causing harm to the development and growth of the new fishery resources and to native fisheries.

In recent years, for example, the release of exotic organisms via ships' ballast water has become a pressing issue, with profound implications for fisheries resources, mariculture, and other activities. These issues are dealt with separately by the Study Group on Ballast and Other Ship Vectors (SGBOSV) and are not considered within this code.

The International Council for the Exploration of the Sea, through its Working Group on Introductions and Transfers of Marine Organisms and in cooperation with other ICES Working Groups and with the European Inland Fisheries Advisory Commission (EIFAC) of the Food and Agriculture Organization of the United Nations (FAO), has addressed these three levels of concern since 1973.

On 10 October 1973, the Council adopted the first version of what was to become an internationally recognized "Code of Practice" on the movement and translocation of non-native species for fisheries enhancement and mariculture purposes. The Code was set forth "to reduce the risks of adverse effects arising from introduction by non-indigenous marine species". Subsequent modifications, proposed by the ICES Working Group on Pathology and Diseases of Marine Organisms in 1978 and by the then newly reconvened ICES Working Group on the Introduction of Non-Indigenous Marine Organisms in 1979, led to the publication of a "Revised Code", adopted by ICES in October 1979. The "1979 Code" became the standard for international policy and the version of the Code most widely used, cited, and translated for the next ten years. Minor revisions and additions over the decade resulted in the adoption in October 1990 of a "1990 Revised Code", followed by the "1994 Code" adopted by ICES in September 1994 (ICES, 1995). The "1994 Code" took into account several updates and included genetic issues for the first time.

The ***2003 Code****, presented here, includes all concerns expressed in the 1994 Code of Practice (ICES, 1995) and follows the precautionary approach adopted from the FAO principles (FAO, 1995), with the goal of reducing the spread of exotic species. It accommodates the risks associated with current commercial practices including trade in ornamental species and bait organisms, research, and the import of live species for immediate human consumption (these are not species that are intended to be released to the environment, so a notification to ICES is neither appropriate nor practical). It also includes species that are utilized to eradicate previously introduced harmful and native species, as well as genetically modified organisms (GMOs). It outlines a consistent, transparent process for the evaluation of a proposed new introduction, including detailed biological background information and an evaluation of risks.*

ICES views the Code of Practice as a guide to recommendations and procedures. As with all Codes, the current Code has evolved with experience and with changing technological developments. This latest version of the Code reflects the past thirty years of experience with the evolution of new fisheries and genetic technologies. While initially designed for the ICES Member Countries concerned with the North Atlantic and adjacent seas, all countries across the globe are encouraged to implement this Code of Practice. Public awareness of the concerns associated with introductions and transfers of marine organisms is essential to assist in the prevention of problems associated with such introductions. Countries are therefore encouraged to ensure the widest distribution of this code.

A brief outline of the ICES Code of Practice 2003

The ICES Code of Practice sets forth recommended procedures and practices to diminish the risks of detrimental effects from the intentional introduction and transfer of marine (including brackish water) organisms. The Code is aimed at a broad audience since it applies to both public (commercial and governmental) and private (including scientific) interests. In short, any persons engaged in activities that could lead to the intentional or accidental release of exotic species should be aware of the procedures covered by the Code of Practice.

The Code is divided into seven sections of recommendations relating to: (I) a strategy for implementation, (II) the steps to take prior to introducing a new species, (III) the steps to take after deciding to proceed with an introduction, (IV) policies for ongoing introductions or transfers which have been an established part of commercial practice, and (V–VI1) the steps to take prior to releasing genetically modified organisms. A section on "Definitions" is included with the Code.

The contents of Sections II–VII have been referred to above and in ICES reports (ICES, 1984, 1988, 1994). Section I provides a strategy for implementation. In recent years, for example, the release of exotic organisms via ships' ballast water has become a pressing issue, with profound implications for fisheries resources, mariculture, and other activities Sections V–VII, dealing with genetically modified organisms (GMOs), have been revised by the Working Group on the Application of Genetics in Fisheries and Mariculture (WGAGFM, 2002).

The Code is presented in a manner that permits broad and flexible application to a wide range of circumstances and requirements in many different countries, while at the same time adhering to a set of basic scientific principles and guidelines.

ICES Member Countries contemplating new introductions are requested to present in good time to the Council a detailed prospectus on the rationale and plans for any new introduction of a marine (brackish) species; the contents of the prospectus are detailed in Section II of the Code and Appendix A (see summary below and www.ices.dk). The Council may then request its Working Group on Introductions and Transfers of Marine Organisms (WGITMO) to consider the prospectus and comment on it. The Working Group, in turn, may request more information before commenting on a proposal. Guidelines to be followed are described, with details in appendices on the ICES website.

If any introduction or transfer proceeds following approval, ICES requests Member Countries to keep the Council informed about it, both through providing details of the broodstock established and the fate of the progeny, and through submitting progress reports after a species is released into the wild. The specifics of this stage are detailed in Section III of the Code.

ICES has published two extended guides to the Code, one in 1984 as Cooperative Research Report No. 130, entitled "Guidelines for Implementing the ICES Code of Practice Concerning Introductions and Transfers of Marine Species", and another in 1988 as Cooperative Research Report No. 159, entitled "Codes of Practice and Manual of Procedures for Consideration of Introductions and Transfers of Marine and Freshwater Organisms". These reports are available in many libraries and from the ICES Secretariat. ICES views the Code of Practice as a guide to recommendations and procedures. As with all Codes, the current Code has evolved with experience and with changing technological developments. The latest (2003) version of the Code reflects the past thirty years of experience with the evolution of new fisheries and genetic technologies.

We are pleased to present the ICES Code of Practice in this fashion for wide consideration, and we welcome advice and comments from both Member Countries and our colleagues throughout the world. Recommendations and suggestions should be directed to the General Secretary of ICES in Copenhagen, Denmark.

Stephan Gollasch

Chair, ICES Working Group on Introductions and Transfers of Marine Organisms

Stig Carlberg

Chair, ICES Advisory Committee on the Marine Environment

M. Møller Hansen

Chair, ICES Working Group for the Application of Genetics in Fisheries and Mariculture

ICES CODE OF PRACTICE ON THE INTRODUCTIONS AND TRANSFERS OF MARINE ORGANISMS 2003

All introductions and transfers of marine organisms carry risks associated with target and non-target species (including disease agents). Once established, introduced species can spread from foci of introductions and have undesirable ecological, genetic, economic, and human health impacts.

Introductions of marine organisms occur in the course of many human activities, including but not limited to aquaculture, stocking, live trade (e.g., species used for aquaria, ornamentals, bait, and food), research, biocontrol, and the use of genetically modified organisms. Even species introduced intentionally into closed systems can be released accidentally. Thus, introductions can result whenever live organisms are moved, regardless of the original intent. As a result, a risk of introduction and subsequent impacts exists with any movement and should be considered explicitly.

This Code of Practice provides a framework to evaluate new intentional introductions, and also recommends procedures for species that are part of current commercial practices to reduce the risk of unwanted introductions, and adverse effects that can arise from species movement.

I Strategy for implementation

a) To protect indigenous as well as previously intentionally introduced species and to meet international obligations (e.g., Convention on Biological Diversity), agencies of Member Countries should fully implement the Code of Practice and apply all regulatory measures possible to prevent unauthorized introductions.

b) To reduce illegal and unauthorized introductions, Member Countries are also encouraged to increase public awareness about the risks associated with importing live products.

c) Countries that are not members of ICES are encouraged to adopt such management measures.

II Recommended procedure for all species prior to reaching a decision regarding new introductions

a) Member Countries contemplating any new introduction are expected to submit to the Council well in advance a detailed prospectus (see Appendix A) on the proposed new introduction(s) for evaluation and comment.

b) The prospectus should include the purpose and objectives of the introduction, the stage(s) in the life cycle proposed for introduction, the native range, the donor location, and the target area(s) of release. The prospectus should also include a review of the biology and ecology of the species as these pertain to the introduction (such as the physical, chemical, and biological requirements for reproduction and growth, and natural and human-mediated dispersal mechanisms) and information on the receiving environment.

c) The prospectus should also provide a detailed analysis of the potential impacts on the aquatic ecosystem of the proposed introduction. This should include, wherever possible, assessments from previous introductions. This analysis should include a thorough review of:

i) the ecological, genetic, and disease impacts and relationships of the proposed introduction in its natural range and donor location;
ii) the expected ecological, genetic, and disease impacts and relationships of the introduction in the proposed release site and projected range, as well as vectors for further distribution;
iii) an economic assessment, where appropriate.

d) The prospectus should conclude with an overall assessment of the issues, problems, and benefits associated with the proposed introduction. An evaluation of risks (see Appendix B) should be included.

e) Upon review of the prospectus, the ICES Council will provide comments and recommendations on the proposed introduction.

III If the decision is taken to proceed with the introduction

a) Using internationally recognized protocols, such as the Office International des Épizooties (OIE), or any other appropriate protocols available at the time, review the health records of the donor location and surrounding area of the organisms to be introduced.

b) The introduced organisms should be used to establish a broodstock for the production of progeny. The organisms should be transferred into a quarantine facility (see Appendix C). This facility should be in the recipient country or other location agreed to by the recipient country.

c) The imported consignment(s) is not to be released to the wild, and should be separated from subsequent progeny.

d) Only progeny of the introduced species may be transplanted into the natural environment, provided that:

i) a risk assessment indicates that the likelihood of negative genetic and environmental impacts is minimal,
ii) no disease agents, parasites, or other non-target species become evident in the progeny to be transplanted, and
iii) no unacceptable economic impact is to be expected.

e) During the pilot phase, the progeny, or other suitable life stages, should be placed on a limited scale into open waters to assess ecological interactions with native species, and especially to test risk assessment assumptions. Contingency plans, including the removal of the introduced species from the environment, should be ready for immediate implementation.

f) A monitoring programme addressing specific issues (see Appendix D) of the introduced species in its new environment should be undertaken, and annual progress reports should be submitted to ICES for review at meetings of the Working Group on Introductions and Transfers of Marine Organisms until the review process is considered complete.

IV Recommended procedure for introduced or transferred species which are part of current commercial practice

a) All products should originate from sources in areas that meet current codes, such as the OIE International Aquatic Animal Health Code or equivalent EU directives.

b) Live products destined for consumption, processing, and aquarium or display should not be placed into the natural environment.

c) For organisms to be released into the natural environment, there should be documented periodic inspections (including microscopic examination) of material prior to exportation to confirm freedom from exotic accompanying (non-target) species including disease agents. If an inspection reveals any undesirable development, it must be immediately reported and importation must be immediately discontinued. Findings and remedial actions should be reported to the International Council for the Exploration of the Sea.

d) If required, there should be inspection, disinfection, quarantine or destruction of the introduced organisms and transfer material (e.g., transport water, packing material, and containers) based on OIE or EU directives.

e) Consider and/or monitor the genetic impact that introductions or transfers have on indigenous and previously introduced species or distinct genetic stocks, to reduce or prevent detrimental changes to genetic diversity.

Note: It is recognized that different countries will have special requirements for the inspection and control of the consignment in the donor and recipient countries.

V General considerations regarding the release of genetically modified organisms (GMOs)

a) Recognizing that little information still exists on the genetic, ecological, and other effects of the release of genetically modified organisms into the natural environment (where such releases may result in the mixing of altered and wild populations of the same species, and in changes to the environment), the Council urges Member Countries to establish strong legal measures[1] to regulate such releases, including the mandatory licensing of physical or juridical persons engaged in genetically modifying, or in importing, using, or releasing any genetically modified organism.

VI Recommended procedure for all GMOs prior to reaching a decision regarding new releases

a) Member Countries contemplating any release of genetically modified organisms into open marine and brackish environments are requested at an early stage to notify the Council about such releases. This notification should include a risk assessment of the effects of this release on the environment and on natural populations.

b) GMO risk assessment should particularly involve consideration of:

i) The genetic and phenotypic characteristics of the modified organism, i.e., both the traits introduced or modified and other secondary phenotypic changes induced by the genetic modification, such as the construction and/or vector employed. The significance of the introduced or modified trait in relation to the biology of the parental organism should be evaluated.
ii) Characteristics of the ecosystems that the GMO might access.
iii) Possible interactions of the GMO with species of the ecosystems that might be accessed, in order to determine whether the release of the GMO poses genetic and/or ecological hazards.

c) If possible, experiments in simulated natural environments are recommended. Such experiments should be conducted using secure systems to prevent escapes of GMOs from the

[1]Such as the European Economic Community "Council Directive of 12 March 2001 on the Deliberate Release into the Environment of Genetically Modified Organisms (2001/18/CE)", Official Journal of European Communities, No. L, 106: 1–39 (2001).

experimental facilities at any life stage. The following points should be particularly assessed and reported:

i) Phenotypic traits associated with the GMO in a simulated natural environment;
ii) The behaviour of transgenic aquatic organisms in a simulated natural environment;
iii) The competitive advantages/disadvantages of transgenic aquatic organisms;
iv) The degree to which transgenic aquatic organisms are capable of mating with a native population, including their reproductive performance in competition with wild conspecifics;
v) The success of that mating as defined by numbers of offspring;
vi) The relative fitness of juveniles of pure transgenic crosses, hybrids between native and transgenic crosses, and the pure native crosses.

VII If the decision is taken to proceed with the release, the following action is recommended:

a) It is recommended that initial releases of transgenic (GMO) organisms be reproductively sterile in order to avoid transfer of the gene construct to wild organisms. However:
   i) Mass production of sterile progeny requires the maintenance of fertile transgenic parental stocks. The risk assessment of these stocks should also be addressed.
   ii) It should be noted that many current sterilization techniques are not 100% efficient and that many aquatic species have very high fecundity.
   iii) Mass releases of sterile organisms could still negatively impact the ecosystem and affect wild populations through competition.

b) Monitoring should be undertaken to ensure that GMOs, due to their nature, do not negatively affect wild populations and ecosystems after their release.

DEFINITIONS

For the application of this Code, the following definitions shall be used.

Aquarium (= ornamental) species

All species imported or transferred into confinement for ornamental indoor or outdoor use.

Bait organisms

Live specimens used (e.g., on a hook or in a trap) to allure target species.

Biocontrol species

The intentional release of an organism that is intended to consume, infect, or debilitate a selected species to decrease its population size. Note: The possible limited specificity of biocontrol species is of concern as native species might be negatively affected.

Broodstock

Specimens of a species in any life stage from which a first or subsequent generation/growth may be produced for possible introduction to the environment.

Current commercial practice

Established and ongoing cultivation, rearing, or placement of an introduced or transferred species in the environment for economic or recreational purposes, which has been ongoing for a number of years.

Disease agent

Any organism, including parasites and prions, that causes or contributes to the development of a disease.

Donor location (= source localities)

Specific localities in a country or zone from which the import or transfer originates.

Genetic diversity

All of the genetic variation in an individual population, or species.

Genetically modified organism (GMO)

An organism in which the genetic material has been altered anthropogenically by means of recombinant DNA technologies. This definition includes transgenic organisms, i.e., an organism bearing within its genome one or more copies of novel genetic constructs produced by recombinant DNA technology, but excludes chromosome manipulated organisms (i.e., polyploids), where the number of chromosomes has been changed through cell manipulation techniques.

Indigenous (= native) species

A species or lower taxon living within its natural range (past or present) including the area which it can reach and occupy using its natural dispersal systems (modified after CBD, GISP).

Introduced species (= non-indigenous species, = exotic species)

Any species transported intentionally or accidentally by a human-mediated vector into aquatic habitats outside its native range. Note: Secondary introductions can be transported by human-mediated or natural vectors.

Marine species

Any aquatic species that does not spend its entire life cycle in fresh water.

Native range

Natural limits of geographical distribution of a species (modified after Zaitsev and Ozturk, 2001).

New introduction

The human-mediated movement of a species outside its present distribution.

Non-target species

Any species inadvertently accompanying in, on, or with the species intended for introduction or transfer.

Progeny

Next generation(s) of an organism. Also included are new stages/fragments of seaweeds, protists, and clonal organisms.

Quarantine

The facility and/or process by which live organisms and any of their accompanying organisms can be held or reared in isolation from the surrounding environment, including sterilization procedures.

Release

Voluntary or accidental dissemination of an organism, or its gametes, outside its controlled area of confinement.

Transferred species (= transplanted species)

Any species intentionally or accidentally transported and released within areas of established populations, and continuing genetic flow where it occurs.

Vector

Any living or non-living carrier that transports living organisms intentionally or unintentionally.

Zone

Part of a coastal area or an estuary of one or more countries with the precise geographical delimitation that consists of a homogeneous hydrological system (modified after OIE).

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OVERVIEW OF APPENDICES TO THE CODE OF PRACTICE

The following provides an overview of the four Appendices referred to in the 2003 version of the ICES Code of Practice on the Introductions and Transfers of Marine Organisms. To ensure that the appendices are current and that the most recent information is included, appendices (with an example of a case study) will only be available on the Internet (www.ices.dk).

Appendix A. Prospectus

This Appendix provides detailed information on suggested guidelines for the prospectus including, but not limited to:

- potential of transfer of disease agents, parasites, and non-target species;
- review of previous introductions of the candidate species.

This information is used to conduct the biological risk assessment (see Appendix B). To be scientifically valid, the information provided needs to be based on a thorough literature review.

The prospectus also needs to include a contingency plan in case immediate eradication of the introduced species needs to be carried out.

The proponent should design an appropriate monitoring programme that will document impacts in the receiving environment.

Appendix B. Risk Assessment

This Appendix provides a detailed, consistent approach for evaluating the risk of genetic, ecological, and disease impacts in the proposed receiving environment, as well as the potential for introducing non-target species. This review should be based in part on the information provided in the Prospectus (see Appendix A).

There will be an assessment of each potential hazard as to the probability of the establishment and consequences of the establishment in the receiving environment. Mitigation factors and management issues will also be reviewed.

The precautionary principle will be taken into account in the final outcome of the risk assessment.

Appendix C. Quarantine

The intention of the quarantine process is to:

- prevent the escapes of target and non-target species into the environment;
- ensure freedom from disease agents in broodstock and progeny prior to release from the quarantine system;
- protect broodstock.

The size of the facility, and the extent of the quarantine measures, will depend on the characteristics of the species being introduced. Quarantine measures may also be required for some species transfers.

The Appendix provides detailed information on suggested requirements for quarantine facilities including, but not limited to:

- transport of broodstock;
- quarantine facilities;
- stock management in isolation;
- record keeping;
- disinfection.

Appendix D. Monitoring

The purpose of the monitoring programme is to assess the impact of the introduced organisms on the environment, ecosystem function, and biodiversity (including genetic biodiversity). The monitoring should be adjusted according to the type of organism and its potential dispersal range. The vectors responsible for further dispersal need to be identified.

Appropriate monitoring should be carried out in phases:

- initial baseline monitoring study before the introduction;
- continuing monitoring subsequent to pilot study release; and
- continuing monitoring following increases in the scale of the project.

The results of the monitoring may be reported to and assessed by WGITMO before the next phase is undertaken. Questions outlined in the Appendix should be addressed as far as possible.

ANNEX 8: RESULTS OF ANALYSES OF MULTIVARIATE BIOLOGICAL COMMUNITY DATA

The results of analyses on benthic macrofauna data using Multidimensional Scaling (MDS) and other techniques are presented here.

Data used

Time-series data on benthic macrofauna from three stations covering the period 1980–2001 were used. The data were obtained from the old HELCOM database stored at ICES and from the National Environmental Research Institute, Denmark. To complement the benthic macrofauna data, environmental data on nutrients, chlorophyll *a,* and oxygen from the same stations were obtained from the ICES oceanographic database.

The stations are situated in an area that sometimes experiences oxygen deficiency during the autumn. The extent and degree of the oxygen deficiency vary from year to year depending primarily on the load of nutrients and the meteorological conditions that year.

Analysis of data on benthic fauna and environmental variables

MDS ordinations were made on both abundance and biomass from the single stations using the Bray-Curtis similarity coefficient. Ordinations were made on untransformed, fourth root transformed, and presence / absence data. This sequence of transformations progressively reduces the influence of the most abundant and highest biomass species on the MDS ordinations.

The environmental data were available at a much higher temporal resolution than the annual macrofauna data. The environmental data were therefore aggregated to give the eight annual environmental variables listed in Table A8.1. The choice of aggregated variables was based on the following assumptions:

1) There is generally a time lag in the response of macrofauna to environmental variables (e.g., available nutrients are taken up by phytoplankton which then sinks to the bottom where it acts as food for the macrofauna);
2) Low oxygen values (which usually occur between August and October) influence the faunal composition during the following year (the fauna samples are usually from the spring);
3) Nutrient concentrations in January–February (when there is virtually no uptake of nutrients by phytoplankton) act as a measure of the potential primary production in the spring phytoplankton bloom.

The eight annual environmental variables were analysed using a Principal Component Analysis (PCA) on a normalized data matrix. The normalization made each environmental variable equally important in forming the principal components.

The ranks of the similarity matrix (Bray-Curtis similarity) for the MDS ordination of the fauna $\{r_i;\ i = 1, \ldots, N\}$ and the ranks of the similarity matrix (Euclidean distance) for the PCA ordination of the environmental variables $\{s_i;\ I = 1,\ldots., N\}$ were compared using a Spearman rank correlation coefficient

$$\rho_s = 1 - \frac{6}{N(N^2-1)} \sum_{i=1}^{N} (r_i - s_i)^2$$

where $N = n(n - 1)/2$ and n is the number of samples (i.e., years). It is tempting to refer ρ_s to standard statistical tables of Spearman's rank correlation to assess whether the two patterns are "significantly" matched. However, this is invalid because the ranks are not mutually independent since they are based on a large number of interdependent similarity calculations. The ρ_s can, however, still be used as an index of agreement of the two matrices.

The Spearman rank correlations for three stations are shown in Table A8.1. The correlations are based on the environmental variables giving the best "match". The match is rather modest in most cases. The relatively poor match might be caused by the influences of other environmental variables not included in this study and/or spatial heterogeneity in the macrofauna.

Taxonomic distinctness of the macrofauna

The average taxonomic distinctness (Δ^+) is defined (Warwick and Clarke, 1995) as:

$$\Delta^+ = [\Sigma\Sigma_{i<j}\ \omega_{ij}] / [S(S-1)2]$$

where the double summation is over all pairs of species i and j ($i,j = 1,2,\ldots S$; $i<j$) and ω_{ij} is the taxonomic distance between species i and j. Since it is difficult to determine the true taxonomic distance between different taxa, Clarke and Warwick (1999) advocated a simple linear scaling (see Figure A8.1). The distance between two species with the greatest taxonomic difference (in this case different phyla) is set to $\omega = 100$, whereas the distance between two individuals of the same species is set to $\omega = 0$.

Table A8.1. The results of comparing the similarity matrices for the macrofauna and the environmental variables at stations 413, 444, and 939. The abundance and biomass of the macrofauna were analysed on untransformed, fourth root transformed, and presence/absence data. The Spearman rank correlations are based on the environmental variables (shown in bold type) that give the best match. For example, the fourth root transformed abundance data from station 413 was best matched (Spearman rank correlation of 0.467) with the environmental variables (1) lowest oxygen value the year before, (5) concentration of chlorophyll *a* the same year, (6) concentration of nitrate and nitrite the year before, (7) concentration of phosphate the year before, and (8) concentration of chlorophyll *a* the year before.

	413		444		939	
	Abundance	Biomass	Abundance	Biomass	Abundance	Biomass
No transformation	0.359 (**2, 6-8**)	0.367 (**1, 6-8**)	0.29 (**1, 2**)	0.314 (**2, 7**)	0.23 (**4**)	0.185 (**4, 7**)
4th root	0.467 (**1, 5-8**)	0.446 (**1,5-8**)	0.244 (**1, 2**)	0.242 (**1, 2**)	0.393 (**4, 7**)	0.403 (**4, 7**)
Presence/absence	0.434 (**1, 5-8**)		0.215 (**1, 2**)		0.406 (**4, 7**)	

1 Lowest oxygen value during August-October (year before)
2 Mean oxygen value during August-October (year before)
3 Concentration of nitrate + nitrite during January-February (same year)
4 Concentration of phosphate during January-February (same year)
5 Concentration of chlorophyll *a* during February-March (same year)
6 Concentration of nitrate + nitrite during January-February (year before)
7 Concentration of phosphate during January-February (year before)
8 Concentration of chlorophyll *a* during February-March (year before)

This is a very intuitive definition of biodiversity, measuring the average breadth of a sample. The taxonomic distinctness of a series of samples can be plotted against the number of species in each sample (Figure A8.2).

The average taxonomic distinctness (Δ^+) is independent of the number of species in a sample, so this makes it possible to compare samples irrespective of sample size. The low taxonomic distinctness in some years (Figure A8.2) could not be related to particular environmental variables, although taxonomic distinctness seemed to be generally below average in years after oxygen deficiencies.

Since the taxonomic distinctness is independent of sample size, it could be useful in the analysis of historical data where sample sizes were not comparable to those of the present. This is particularly important when defining baseline conditions (no or very little human impact) for marine biological communities. This reference condition is a basis for several international monitoring programmes (e.g., the Water Framework Directive).

Summary

The ACME noted that many ordination techniques such as MDS are well suited for detailed analyses of specific data sets, where local expertise can be used to guide both the data analysis and the interpretation of the results. The challenge is to turn these techniques into tools that can be used in routine monitoring assessments. The ACME recommends that multidimensional scaling and other ordination and multivariate techniques should be further explored to produce standard data products for use in routine monitoring assessments of biological community data.

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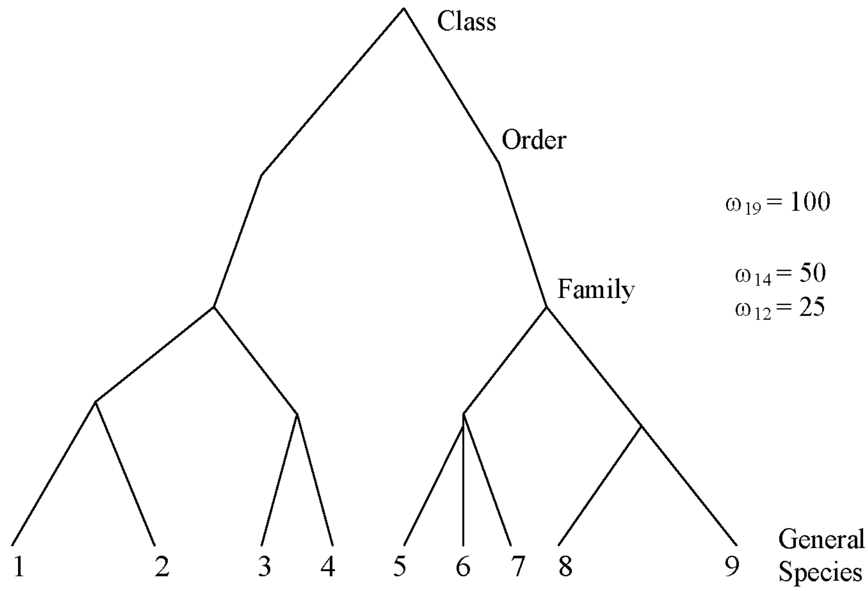

Figure A8.1. An example of how to calculate ω in a simple taxonomic tree with ı

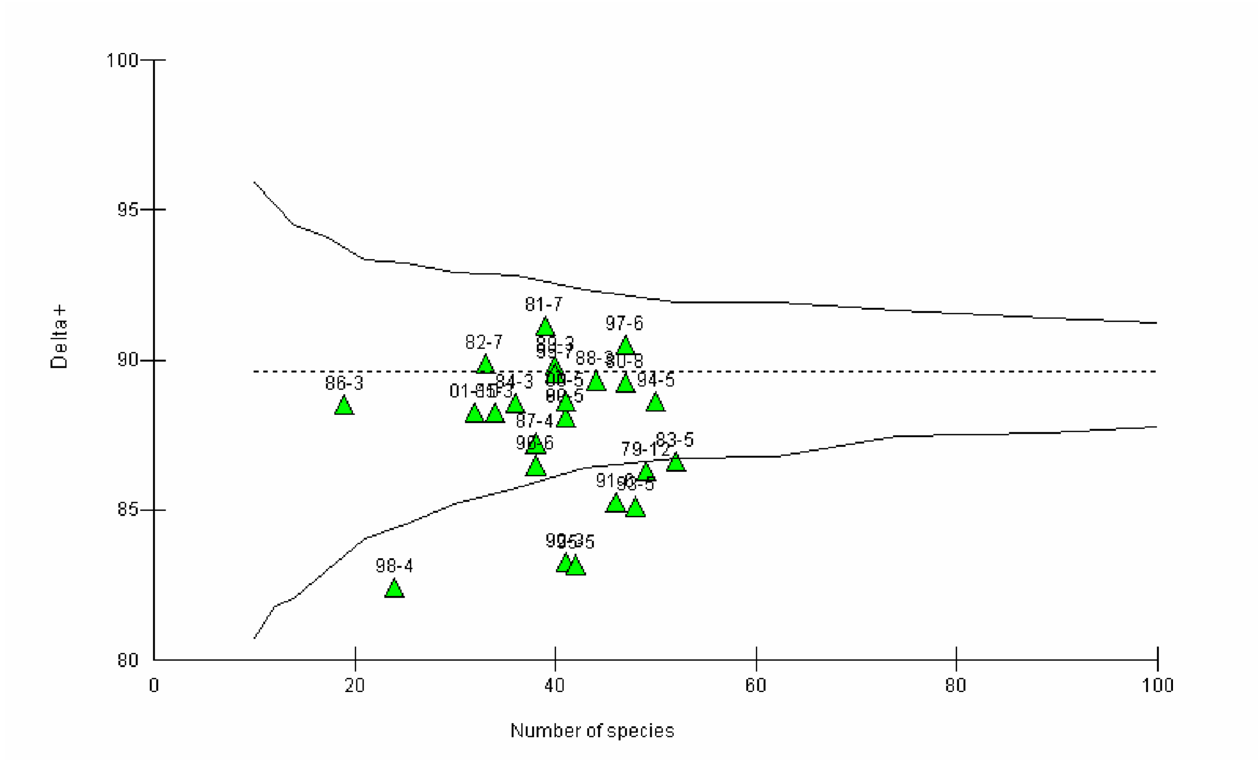

Figure A8.2. The average taxonomic distinctness (Δ^+) (dotted line) based on the total list of species for the area and the actual taxonomic distinctness of samples from a single station in the Sound. The plotting symbol denotes the year and month of the sample (e.g., 98-4 is April 1998). The solid lines indicate limits within which 95% of simulated Δ^+ values lie.

ANNEX 9: IMPLICATIONS OF THE WATER FRAMEWORK DIRECTIVE FOR MARICULTURE OPERATIONS

BACKGROUND

The EU Water Framework Directive 2000/60/EC (WFD) will be the primary EU driver for the improvement of groundwater and surface water quality over the next decades. Under the Directive, definitions of good ecological quality will be agreed for all types of water bodies, covering all surface waters (and groundwater) in the EU. Good ecological quality will then be the target for improvement measures to be adopted by member states and their environmental agencies.

Aquaculture is not specifically mentioned in the Directive. However, mariculture will be viewed as being a source of environmental pressures with the potential to adversely affect primary indices of ecological quality in the transitional and coastal water bodies where mariculture operations are located. As such, it is likely that such areas will be subject to operational monitoring, as defined under the WFD.

The implementation of the WFD is a major exercise and the timetable for its completion stretches ahead during the first two decades of the 21st century, towards the target of achieving "good" ecological status in all water bodies by 2015. In recognition of this, the EU established a Common Implementation Strategy, which consisted of a series of international projects addressing particular processes within the overall implementation. These groups have now prepared their reports, and it is now possible to make preliminary analyses to identify the areas where mariculture activities, discharges, etc., may come within the scope of WFD.

In order to do this, it is first necessary to describe the general processes involved in the implementation of the WFD. The following text has largely been derived from the output documents of the EU Common Implementation Strategy projects, particularly those concerned with Coastal and Transitional waters (project 2.4) and Monitoring (Project 2.7).

Purpose of the Directive

The Directive establishes a framework for the protection of all waters (including inland surface waters, transitional waters, coastal waters, and groundwater) which:

1) Prevents further deterioration of, protects, and enhances the status of water resources;
2) Promotes sustainable water use based on long-term protection of water resources;
3) Aims at enhancing protection and improvement of the aquatic environment through specific measures for the progressive reduction of discharges, emissions, and losses of priority substances and the cessation or phasing-out of discharges, emissions, and losses of the priority hazardous substances;
4) Ensures the progressive reduction of pollution of groundwater and prevents its further pollution;
5) Contributes to mitigating the effects of floods and droughts.

The concept central to the WFD is *integration,* which is seen as key to the management of water protection within the river basin district:

1. **Integration of environmental objectives**, combining quality, ecological, and quantity objectives for protecting highly valuable aquatic ecosystems and ensuring a general good status of other waters;
2. **Integration of all water resources**, combining fresh surface water and groundwater bodies, wetlands, and coastal water resources **at the river basin scale**;
3. **Integration of all water uses, functions, and values** into a common policy framework, i.e., investigating water for the environment, water for health and human consumption, water for economic sectors, transport, leisure, water as a social good;
4. **Integration of disciplines, analyses and expertise,** combining hydrology, hydraulics, ecology, chemistry, soil sciences, technology, engineering, and economics to assess current pressures and impacts on water resources and identify measures for achieving the environmental objectives of the Directive in the most cost-effective manner;
5. **Integration of water legislation into a common and coherent framework.** The requirements of some old water legislation (e.g., the Freshwater Fish Directive) have been reformulated in the WFD to meet modern ecological thinking. After a transitional period, these old Directives will be repealed. Other pieces of legislation (e.g., the Nitrates Directive and the Urban Wastewater Treatment Directive) must be coordinated in river basin management plans where they form the basis of the programmes of measures;
6. **Integration of all significant management and ecological aspects** relevant to sustainable river basin planning including those which are beyond the scope of the Water Framework Directive such as flood protection and prevention;
7. **Integration of a wide range of measures, including pricing and economic and financial instruments, in a common management approach** for achieving the environmental objectives of the Directive. Programmes of measures are defined in **River Basin Management Plans** developed for each river basin district;

8. **Integration of stakeholders and the civil society in decision making,** by promoting transparency and information to the public, and by offering an unique opportunity for involving stakeholders in the development of river basin management plans;
9. **Integration of different decision-making levels that influence water resources and water status,** whether local, regional or national, for effective management of all waters;
10. **Integration of water management from different Member States,** for river basins shared by several countries, existing and/or future Member States of the European Union.

Actions required under the Directive

The main activities required of Member States are:

a) To identify the individual river basins lying within their national territory and assign them to River Basin Districts (RBDs) and identify competent authorities by 2003 (*Article 3, Article 24*);

b) To characterize river basin districts in terms of pressures, impacts and economics of water uses, including a register of protected areas lying within the river basin district, by 2004 (*Article 5, Article 6, Annex II, Annex III*);

c) To carry out, jointly and together with the European Commission, the intercalibration of the ecological status classification systems by 2006 (*Article 2 (22), Annex V*);

d) To make operational the monitoring networks by 2006 (*Article 8*);

e) Based on sound monitoring and the analysis of the characteristics of the river basin, to identify, by 2009, a programme of measures for achieving cost-effectively the environmental objectives of the WFD (*Article 11, Annex III*);

f) To produce and publish River Basin Management Plans (RBMPs) for each RBD including the designation of heavily modified water bodies, by 2009 (*Article 13, Article 4.3*);

g) To implement water pricing policies that enhance the sustainability of water resources by 2010 (*Article 9*);

h) To make the measures of the programme operational by 2012 (*Article 11*);

i) To implement the programmes of measures and achieve the environmental objectives by 2015 (*Article 4*)

This sequence of activities is summarized in Figure A9.1. The following text will look at some of these processes in more detail.

Defining surface water bodies within transitional and coastal waters

The WFD requires that all surface waters are assigned to river basin districts (RBDs), which will be the primary management units. Within RBDs, surface waters will be allocated into water bodies. These will be the primary units for monitoring, classification, and subsequent improvement (if necessary).

The Directive requires surface waters within the River Basin District to be split into water bodies (Figure A9.1). Water bodies represent the classification and management unit of the Directive. A range of factors will determine the identification of water bodies. Some of these will be determined by the requirements of the Directive and others by practical water management considerations. In particular, the definition of water bodies has to take account of the particular pressures that may impact the ecological quality of surface waters in the area.

In deriving an appropriate system of water bodies, the Directive only requires sub-divisions of surface water that are necessary for the clear, consistent, and effective application of its objectives. Sub-divisions of coastal and transitional waters into smaller and smaller water bodies that do not support this purpose should be avoided.

For example, the need to keep separate two or more contiguous water bodies of the same type depends upon the pressures and resulting impacts. For example, a discharge may cause organic enrichment in one water body but not in the other. Such an area of one type could therefore be divided into two separate water bodies with different classifications. If there were no impact from the discharge, it would not be necessary to divide the area into two water bodies as it would have the same classification and should be managed as one entity.

According to Annex II of the Directive, Member States shall assign surface water bodies to one of the following categories: rivers, lakes, transitional, coastal, artificial or heavily modified surface water bodies. These categories must then be further divided into types. The Directive indicates that Types can be defined using one of two systems (A and B). Most Member States have expressed the opinion that system B will be applied. This is because the differences in biological compositions and community structures normally depend on more descriptors than the very limited range included in system A.

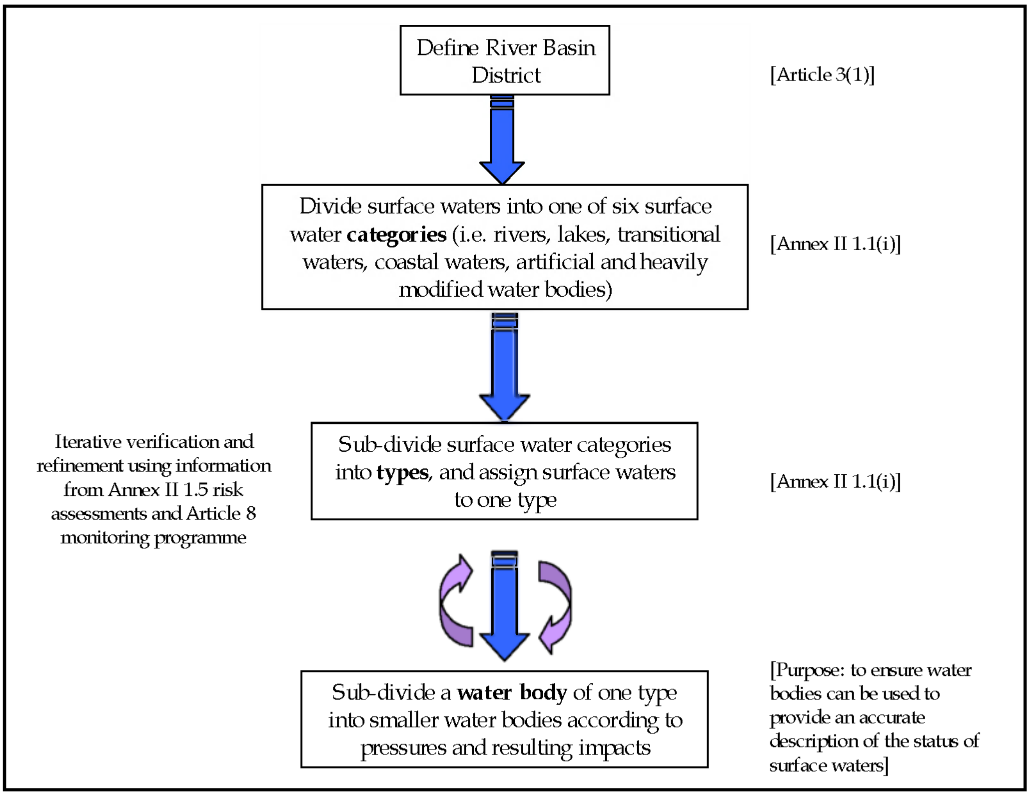

Figure A9.1. Summary of suggested hierarchical approach to the identification of surface water bodies.

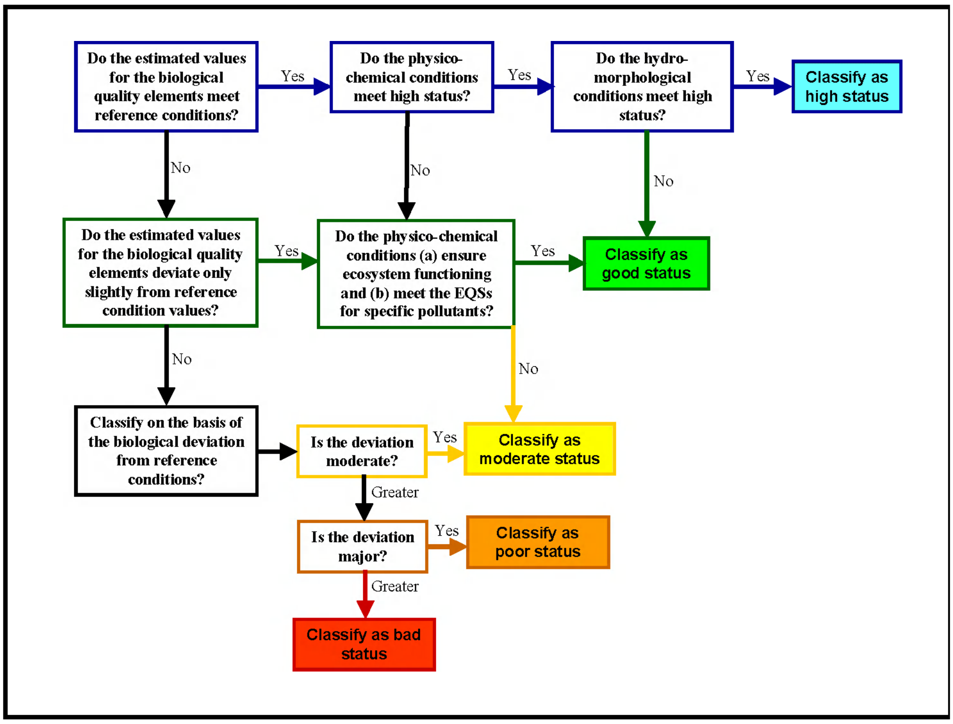

Figure A9.2. Indication of the relative roles of biological, hydromorphological, and physico-chemical quality elements in ecological status classification according to the normative definitions in Annex V 1.2. A more detailed understanding of the role of physico-chemical parameters in the classification of the ecological status will be developed in specific guidance on this issue during 2003.

The factors listed in Annex II of the Directive for defining typologies for coastal and transitional waters under System B are as follows:

Transitional Waters

Alternative Characterization	Physical and chemical factors that determine the characteristics of the transitional water and hence the biological population structure and composition
Obligatory factors	latitude longitude tidal range salinity
Optional factors	depth current velocity wave exposure residence time mean water temperature mixing characteristics turbidity mean substratum composition shape water temperature range

Coastal Waters

Alternative Characterization	Physical and chemical factors that determine the characteristics of the coastal water and hence the biological population structure and composition
Obligatory factors	latitude longitude tidal range salinity
Optional factors	current velocity wave exposure mean water temperature mixing characteristics turbidity retention time (of enclosed bays) mean substratum composition water temperature range

Classification

Once typology has been established, water bodies are then assigned to types. Each water body must be assigned to only one type. The ecological quality of each water body is then assessed against the reference condition defined for that particular type. The reference condition is a description of the biological quality elements that exist, or would exist, at high status, that is, with no, or very minor, disturbance from human activities. The objective of setting reference condition standards is to enable the assessment of ecological quality against these standards.

In defining biological reference conditions, criteria for the physico-chemical and hydromorphological quality elements at high status must also be established. The reference condition is a description of the biological quality elements only. High ecological status incorporates the biological, physico-chemical, and hydromorphological elements.

The classification of ecological status is based upon the status of the biological, hydromorphological, and physico-chemical quality elements (Figure A9.2). The quality elements to be included in classification of transitional and coastal waters are listed in Annex V 1.1.3. and 1.1.4, and reproduced below. The hydromorphological and physico-chemical elements are also referred to as the supporting elements.

Annex V 1.1.3. Transitional Waters	**Annex V 1.1.4. Coastal Waters**
Biological elements:	
• *Composition, abundance and biomass of phytoplankton* • *Composition and abundance of other aquatic flora* • *Composition and abundance of benthic invertebrate fauna* • *Composition and abundance of fish fauna*	• *Composition, abundance and biomass of phytoplankton* • *Composition and abundance of other aquatic flora* • *Composition and abundance of benthic invertebrate fauna*
Hydromorphological elements supporting the biological elements:	
Morphological conditions: • *depth variation* • *quantity, structure and substrate of the bed* • *structure of the inter-tidal zone* *Tidal regime:* • *freshwater flow* • *wave exposure*	*Morphological conditions:* • *depth variation* • *structure and substrate of the coastal bed* • *structure of the inter-tidal zone* *Tidal regime:* • *direction of dominant currents* • *wave exposure*
Chemical and physio-chemical elements supporting the biological elements:	
General: • *Transparency* • *Thermal conditions* • *Salinity* • *Oxygenation conditions* • *Nutrient conditions* *Specific Pollutants:* • *Pollution by all priority substances identified as being discharged into the body of water* • *Pollution of other substances identified as being discharged in significant quantities into the body of water.*	*General:* • *Transparency* • *Thermal conditions* • *Salinity* • *Oxygenation conditions* • *Nutrient conditions* *Specific Pollutants:* • *Pollution by all priority substances identified as being discharged into the body of water* • *Pollution of other substances identified as being discharged in significant quantities into the body of water.*

The Directive definitions of the biological elements at high status are shown below:

Biological elements at high status in transitional waters, taken from Annex V Table 1.2.3.

Element	High Status
Biological Quality Elements	
Phytoplankton	The composition and abundance of the phytoplanktonic taxa are consistent with undisturbed conditions. The average phytoplankton biomass is consistent with the type-specific physico-chemical conditions and is not such as to significantly alter the type-specific transparency conditions. Planktonic blooms occur at a frequency and intensity which is consistent with the type-specific physico-chemical conditions.
Macroalgae	The composition of macroalgal taxa is consistent with undisturbed conditions. There are no detectable changes in macroalgal cover due to anthropogenic activities.
Angiosperms	The taxonomic composition corresponds totally or nearly totally to undisturbed conditions. There are no detectable changes in angiosperm abundance due to anthropogenic activities
Benthic Invertebrate Fauna	The level of diversity and abundance of invertebrate taxa is within the range normally associated with undisturbed conditions. All the disturbance-sensitive taxa associated with undisturbed conditions are present.
Fish Fauna	Species composition and abundance is consistent with undisturbed conditions.

Biological elements at high status in coastal waters, taken from Annex V Table 1.2.4.

Element	High Status
Biological Quality Elements	
Phytoplankton	The composition and abundance of the phytoplanktonic taxa are consistent with undisturbed conditions. The average phytoplankton biomass is consistent with the type-specific physico-chemical conditions and is not such as to significantly alter the type-specific transparency conditions. Planktonic blooms occur at a frequency and intensity which is consistent with the type-specific physico-chemical conditions.
Macroalgae and Angiosperms	All disturbance-sensitive macroalgal and angiosperm taxa associated with undisturbed conditions are present. The levels of macroalgal cover and angiosperm abundance are consistent with undisturbed conditions.
Benthic Invertebrate Fauna	The level of diversity and abundance of invertebrate taxa is within the range normally associated with undisturbed conditions. All the disturbance-sensitive taxa associated with undisturbed conditions are present.

Table A9.2. Priority substances identified under the Water Framework Directive.

PRIORITY SUBSTANCES		
Priority Hazardous Substances	**Priority substances subject to review to priority hazardous substances**	**Priority substances**
1. Brominated diphenylether (only pentabromodiphenylether);	1. Anthracene;	1. Alachlor;
2. Cadmium;	2. Atrazine;	2. Benzene;
3. Chloroalkanes, C10–13;	3. Chlorpyrifos;	3. Chlorfenvinphos;
4. Hexachlorobenzene;	4. Di (ethylhexyl) phthalate (DEHP);	4. Dichloromethane;
5. Hexachlorobutadiene;	5. Diuron;	5. 1,2-Dichloroethane;
6. Hexachlorocyclohexane;	6. Endosulfan;	6. Fluoranthene;
7. Mercury and its compounds;	7. Isoproturon	7. Nickel and its compounds;
8. Nonylphenols;	8. Lead and its compounds;	8. Trichloromethane.
9. Polycyclic Aromatic Hydrocarbons (PAHs);	9. Naphthalene;	
10. Pentachlorobenzene;	10. Octylphenols;	
11. Tributyltin compounds.	11. Pentachlorophenol;	
	12. Simazine;	
	13. Trichlorobenzenes;	
	14. Trifluralin.	

Physico-chemical elements

There are two groups of "chemicals" listed in the WFD. The first is Priority Hazardous Substances, and these are listed in Table A9.2. The Directive requires that Member States aim to cease releases of priority hazardous substances into the environment.

The second group of "chemicals" is listed in Annex VIII of the Directive, and is tabulated below. They are commonly referred to as the "Specific pollutant" physico-chemical elements. These physico-chemical elements are used to describe water bodies and include specific pollutants which are being discharged into the water bodies. They include non-synthetic and synthetic substances, as well as more general water quality parameters.

“Specific pollutant” physico-chemical elements (Annex VIII)

1. Organohalogen compounds and substances which may form such compounds in the aquatic environment.
2. Organophosphorus compounds.
3. Organotin compounds.
4. Substances and preparations, or the breakdown products of such, which have been proved to possess carcinogenic or mutagenic properties or properties which may affect steroidogenic, thyroid, reproduction or other endocrine-related functions in or via the aquatic environment
5. Persistent hydrocarbons and persistent and bio-accumulable organic toxic substances.
6. Cyanides.
7. Metals and their compounds.
8. Arsenic and its compounds.
9. Biocides and plant protection products.
10. Materials in suspension.
11. Substances which contribute to eutrophication (in particular, nitrates and phosphates).
12. Substances which have an unfavourable influence on the oxygen balance (and can be measured using parameters such as BOD, COD, etc.).

Table A9.3. Normative definitions of physico-chemical elements for transitional and coastal waters.

High status	Good status	Moderate status
General conditions:		
The physico-chemical elements correspond totally or nearly totally to undisturbed conditions. Nutrient concentrations remain within the range normally associated with undisturbed conditions. Temperature, oxygen balance, and transparency do not show signs of anthropogenic disturbance and remain within the ranges normally associated with undisturbed conditions.	Temperature, oxygen conditions, and transparency do not reach levels outside the ranges established so as to ensure the functioning of the ecosystem and the achievement of the values specified above for the biological quality elements. Nutrient concentrations do not exceed the levels established so as to ensure the functioning of the ecosystem and the achievement of the values specified above for the biological quality elements.	Conditions consistent with the achievement of the values specified above for the biological quality elements.
Specific synthetic pollutants:		
Concentrations close to zero and at least below the limits of detection of the most advanced analytical techniques in general use.	Concentrations not in excess of the standards set in accordance with the procedure detailed in section 1.2.6 without prejudice to Directive 91/414/EC and Directive 98/8/EC. (<environmental quality standard).	Conditions consistent with the achievement of the values specified above for the biological quality elements.
Specific non-synthetic pollutants:		
Concentrations remain within the range normally associated with undisturbed conditions (background levels).	Concentrations not in excess of the standards set in accordance with the procedure detailed in section 1.2.6. without prejudice to Directive 91/414/EC and Directive 98/8/EC. (<environmental quality standard).	Conditions consistent with the achievement of the values specified above for the biological quality elements.

For the purposes of classification of the physico-chemical quality elements in transitional and coastal waters, the WFD requires that the following are included:

General:

- Transparency
- Thermal conditions
- Oxygen conditions
- Salinity
- Nutrient conditions

Specific Pollutants:

- Pollution by all priority substances identified as being discharged into the body of water
- Pollution of other substances identified as being discharged in significant quantities into the body of water

It is understood that the word "specific" should be taken to indicate that not all the elements listed in Annex VIII need be assessed in each water body, rather that classification should consider those elements which the preceding pressures assessment has indicated to be potentially significant.

The WFD also presents normative definitions of the physico-chemical elements at high, good, and moderate status for **transitional** and **coastal waters,** as shown in Table A9.3 (Annex V, 1.2.3, 1.2.4).

It should be noted that the general area of the definition of EQS values, appropriate analytical techniques, quality assurance, etc., is the subject of continuing debate. Key definitions (e.g., of "close to zero") have not been agreed. These issues are being examined by a sub-group of the Expert Advisory Forum on Priority Substances (EAF PS) dealing with Analysis and Monitoring (AMPS). It has been recommended that the approaches adopted by the EAF PS, AMPS group, be adopted for substances for which national detection limits and background concentrations require to be set.

Monitoring requirements for the Directive

Article 8 of the Directive establishes the requirements for the monitoring of surface water status (and also groundwater status and protected areas). Monitoring programmes are required to establish a coherent and comprehensive overview of water status within each river basin district. The programmes have to be operational at the latest by 22 December 2006, and must be in accordance with the requirements of Annex V.

Annex V indicates that monitoring information from **surface waters** is required for:

1) The classification of status. Supplementing and validating the Annex II risk assessment procedure;
2) The efficient and effective design of future monitoring programmes;
3) The assessment of long-term changes in natural conditions;
4) The assessment of long-term changes resulting from widespread anthropogenic activity;
5) Estimating pollutant loads transferred across international boundaries or discharging into seas;
6) Assessing changes in status of those bodies identified as being at risk in response to the application of measures for improvement or prevention of deterioration;
7) Ascertaining causes of water bodies failing to achieve environmental objectives where the reason for failure has not been identified;
8) Ascertaining the magnitude and impacts of accidental pollution;
9) Use in the intercalibration exercise;
10) Assessing compliance with the standards and objectives of Protected Areas; and
11) Quantifying reference conditions (where they exist) for surface water bodies.

Three types of monitoring for surface waters are described in Annex V: surveillance, operational, and investigative monitoring. These types are to be supplemented by monitoring programmes required for Protected Areas registered under Article 6. Annex V only describes requirements for Drinking Water Protected Areas in surface water and for Protected Areas for habitats and species. Member States may wish to integrate monitoring programmes established for other Protected Areas within the programmes established under the Directive. This is likely to improve the cost-effectiveness of the various programmes.

The text above has described the process by which surface water bodies are identified, categorized, and then assigned to types. Type-specific reference conditions have to be identified for each surface water body type. It is the type-specific reference conditions from each surface water body type that the monitoring results will be compared with to give an assessment of the status of a water body categorized in the water body type. Information on the type and magnitude of the significant anthropogenic pressures to which the surface water bodies in each river basin district are subject has to be collected and maintained. There must then be an assessment of the susceptibility of the surface water status of bodies to the pressures identified, and of the likelihood that surface water bodies within the river basin district will fail to meet the environmental quality objectives set under Article 4. This assessment will use any available existing monitoring data. Also expert judgement and/or modelling approaches (i.e., risk assessment) can be used. For the first assessment, data will not be available from Article 8 monitoring programmes, as they do not have to be operational until the end of 2006: data should be available for subsequent assessments for future River Basin Management Plans (RBMPs). However, many countries already have extensive monitoring programmes. The general relationship between the requirements of Articles 5 and 8 in relation to monitoring are summarized in Figure A9.3.

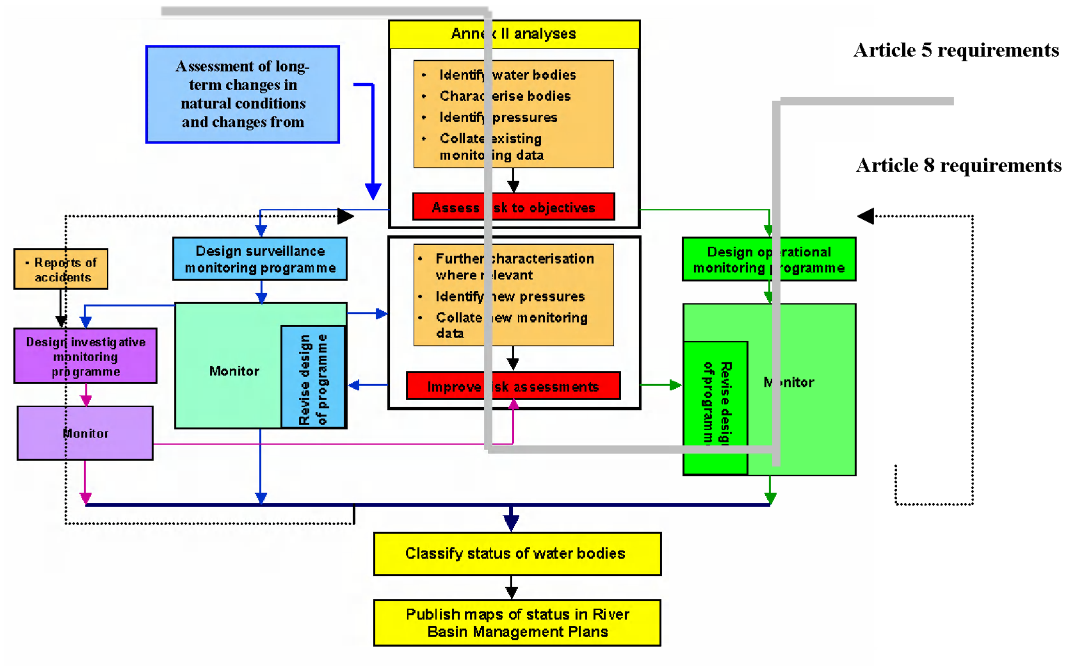

Figure A9.3. Schematic diagram illustrating the relationship between Article 5 and Article 8 in the design of surface water monitoring programmes.

Specifically, fish farms will probably be assessed as potentially affecting the quality of the benthic fauna, the phytoplankton and angiosperm communities, and also hydrochemical conditions such as nutrient and dissolved oxygen concentrations.

However, the position of aquaculture within WFD also needs to be viewed from a different perspective. Successful aquaculture requires good water quality. As the targets for improvements in water quality will be defined within the WFD system, it is important to assess the relationships between the WFD targets and the requirements of aquaculture.

What water bodies should be monitored

The Water Framework Directive covers all waters including inland waters (surface water and groundwater) and transitional and coastal waters up to one sea mile (and for the chemical status also territorial waters which may extend to 12 sea miles) from the territorial baseline of a Member State, independent of the size and the characteristics. For the purpose of the implementation of the Directive, the totality of waters is attributed to geographical or administrative units, in particular the river basin, the river basin district, and the "water body". In addition, groundwater and stretches of coastal waters must be associated with a river basin (district).

Whereas the river basin is the geographical area related to the hydrological system, the river basin district must be designated by the Member States in accordance with the Directive as the "main unit for management of river basins".

One key purpose of the Directive is to prevent further deterioration of, and protect and enhance the status of, aquatic ecosystems, and with regard to their water needs, terrestrial ecosystems and wetlands directly depending on the aquatic ecosystems. The success of the Directive in achieving this purpose and its related objectives will be mainly measured by the status of "water bodies". "Water bodies" are therefore the units that will be used for reporting and assessing compliance with the Directive's principal environmental objectives.

Monitoring is a cross-cutting activity within the Directive and as such there are important interrelationships with other Articles and Annexes of the Directive. A key Article in relation to monitoring and the design of appropriate programmes for surface waters and groundwater is Article 5. Article 5 requires river basin districts to be characterized and the environmental impact of human activities to be reviewed in accordance with Annex II.

Annex II describes a process by which surface water bodies are identified, categorized, and then typified according to one of two systems A or B given in section 1.2 of the Annex. Type-specific reference conditions have to be identified for each surface water body type. It is the type-specific reference conditions from each surface water body type that the monitoring results will be compared with to give an assessment of the status of a water body categorized in the water body type. Information on the type and magnitude of the significant anthropogenic pressures to which the surface water bodies in each river basin district are subject has to be collected and maintained. There must then be an assessment of the susceptibility of the surface water status of bodies to the pressures identified, and of the likelihood that surface water bodies within the river basin district will fail to meet the environmental quality objectives set under Article 4. This assessment will use any available existing monitoring data: the extent of existing data will vary greatly from country to country. Also an expert judgement and/or modelling approach (i.e., risk assessment) can be used. For the first assessment, there will not be data arising from the Article 8 monitoring programmes as they do not have to be operational until the end of 2006; data should be available for subsequent assessments for future RBMPs. Many countries already have extensive monitoring programmes.

The objective of monitoring is to establish a coherent and comprehensive overview of water status within each River Basin District and must permit the classification of all surface water bodies into one of five classes and groundwater into one of two classes. However, this does not mean that monitoring stations will be needed in each and every water body. Member States will have to ensure that enough individual water bodies of each water body type are monitored. They will also have to determine how many stations are required in each individual water body to determine its ecological and chemical status. This process of selecting water bodies and monitoring stations should entail statistical assessment techniques, and should ensure that the overview of water status has an acceptable level of confidence and precision.

For surface water bodies, the Directive requires that sufficient surface water bodies are required to be monitored in a surveillance programme to provide an assessment of the overall surface water status within each catchment and sub-catchment within the river basin district. Operational monitoring is to establish the status of those water bodies identified as being at risk of failing their environmental objectives, and to assess any changes in their status from the programmes of measures. Operational monitoring programmes must use parameters indicative of the quality element or elements most sensitive to the pressure or pressures to which the body or group of bodies is subject. This means that the least number of estimated quality element values may be used in status classification.

The scale of monitoring programmes will be dependent to some degree on the numbers of water bodies – or more accurately on the extent of, and variability in, impacts on the water environment. However, the amount of monitoring required will also depend on the degree to which the characteristics of, and range of pressures on, a Member State's water bodies allow them to be grouped for monitoring purposes.

Article 2.10 of the Directive provides the following definition of a body of surface water: *"Body of surface water" means a* discrete and significant element *of surface water such as a lake, a reservoir, a stream, river or canal, part of a stream, river or canal, a transitional water or a stretch of coastal water.*

Surveillance monitoring of surface waters

Objectives and timing

The objectives of surveillance monitoring of surface waters are to provide information for:

1) Supplementing and validating the impact assessment procedure detailed in Annex II;
2) The efficient and effective design of future monitoring programmes;
3) The assessment of long-term changes in natural conditions; and
4) The assessment of long-term changes resulting from widespread anthropogenic activity.

The results of such monitoring should be reviewed and used, in combination with the impact assessment procedure described in Annex II, to determine requirements for monitoring programmes in the current and subsequent River Basin Management Plans (RBMP).

As has already been described, there will be no information arising from surveillance monitoring for the first risk assessment undertaken under Article 5; monitoring programmes have to be operational by December 2006, and the first Article 5 characterization/risk assessment completed by December 2004. However, any existing monitoring data should be used in the assessment. Many countries have already established extensive monitoring programmes, including monitoring programmes at fish farm locations.

Surveillance monitoring has to be undertaken for at least a period of one year during the period of a RBMP. The deadline for the first RBMP is 22 December 2009. The monitoring programmes must start by 22 December 2006. The first results will be needed for the first draft RBMP to be published at the end of 2008, and then for the finalized RBMPs at the end of 2009. These plans must include status maps.

Surveillance monitoring is also required to provide information on long-term natural changes and long-term changes resulting from widespread anthropogenic activity. Information on the first will be important if such changes are likely to affect reference conditions. Monitoring for long-term natural changes is likely to be focused on high and maybe good status water bodies. This is because such changes (possibly relatively small and gradual) are more likely to be detectable in the absence of the impact of anthropogenic activities which may mask natural changes. In terms of changes resulting from widespread anthropogenic activity, monitoring will be important to determine or confirm the impact of, for example, long-range transport and deposition of pollutants from the atmosphere. If this is likely to lead to a risk of water bodies deteriorating in status (any status level down to poor), then those water bodies or groups of bodies will have to be included in operational monitoring programmes.

Selection of quality elements

For surveillance monitoring, Member States must monitor at least for a period of a year parameters indicative of all biological, hydromorphological, and general physico-chemical quality elements. The relevant quality elements for each type of water are given in Annex V of the Directive. Thus for rivers, the biological parameters chosen to be indicative of the status of each biological element such as the aquatic flora, macro-invertebrates, and fish must be monitored for. For example, in the case of the aquatic flora, the parameters might be presence or absence of indicator species or the population structure. The Directive indicates that monitoring of the biological quality elements must be at an appropriate taxonomic level to achieve adequate confidence and precision in the classification of the quality elements. This applies equally to the three types of surface water monitoring.

Those priority list substances discharged into the river basin or sub-basins must be monitored. Other pollutants (Annex VIII) also need to be monitored if they are discharged in significant quantities in the river basin or sub-basin. No definition, however, of "significance" is given but quantities that could compromise the achievement of one of the Directive's objectives are clearly significant, and as examples, one might assume that a discharge that impacted a Protected Area, or caused exceedence of any national standard set under Annex V 1.2.6 of the Directive or caused a biological or ecotoxicological effect in a water body would be expected to be significant.

A structured approach should be used to inform the process of selecting which chemical should be monitored for in the surveillance monitoring programme. This should be based on a combination of knowledge of use patterns (quantity and locations), pathways for inputs (diffuse and/or point source), and existing information on potential ecological impacts.

Additionally, the selection should be informed by information on the ecological status where indications of toxic impacts are found or from ecotoxicological evidence. This will help to identify situations where unknown chemicals are entering the environment which need investigative monitoring.

Operational monitoring of surface waters

Objectives

The objectives of operational monitoring are to:

- Establish the status of those bodies identified as being at risk of failing to meet their environmental objectives, and
- Assess any changes in the status of such bodies resulting from the programmes of measures.

Operational monitoring (or in some cases investigative monitoring) will be used to establish or confirm the status of bodies thought to be at risk. It is highly focused on parameters indicative of the quality elements most sensitive to the pressures to which the water body or bodies are subject.

Selection of monitoring sites

Operational monitoring has to be undertaken for all water bodies that have been identified, by the review of the environmental impact of human activities (Annex II) and/or from the results of the surveillance monitoring, as being at risk of failing the relevant environmental objectives under Article 4. Monitoring must also be carried out for all bodies into which priority substances are discharged. This implies that monitoring in all such bodies will not necessarily be required as the Directive allows similar water bodies to be grouped and representatively monitored.

In addition, monitoring sites for those priority list substances with environmental quality standards should be selected according to the requirements of the legislation establishing the standards.

For diffuse sources, the selected water bodies need to be representative of the relative risks of the occurrence of the diffuse source pressures, and of the relative risks of the failure to achieve good surface water status. However, in selecting the representative water bodies for operational monitoring it should be taken into account that water bodies can only be grouped, for example, where the ecological conditions are similar or almost similar in terms of the magnitude and type of pressure as well as in terms of hydrological and biological conditions such as retention time and food web structure. In all cases, grouping must be technically or scientifically justifiable.

Selection of quality elements

For operational monitoring, Member States are required to monitor for those biological and hydromorphological quality elements most sensitive to the pressures to which the body or bodies are subject. For example, if organic pollution is a significant pressure on a river, then benthic invertebrates might be the most sensitive and appropriate indicator of that pressure. Thus, in the absence of other pressures, aquatic flora and fish populations may not need to be monitored in those bodies of water. However, the monitoring and assessment system must still be based on the concept of ecological status and not just reflect degrees of organic pollution without comparison to the appropriate reference conditions. This is because its ecological status must be defined.

Investigative monitoring

Investigative monitoring may also be required in specified cases. These are given as:

- where the reasons for any exceedance of Environmental Objectives are unknown;
- where surveillance monitoring indicates that the objectives set under Article 4 for a body of water are not likely to be achieved and operational monitoring has not already been established, in order to ascertain the causes of a water body or water bodies failing to achieve the environmental objectives; or
- to ascertain the magnitude and impacts of accidental pollution.

The results of the monitoring would then be used to inform the establishment of a programme of measures for the achievement of the environmental objectives and specific measures necessary to remedy the effects of accidental pollution.

Investigative monitoring will thus be designed to the specific case or problem being investigated. In some cases, it will be more intensive in terms of monitoring frequencies and focused on particular water bodies or parts of water bodies, and on relevant quality elements. Ecotoxicological monitoring and assessment methods would in some cases be appropriate for investigative monitoring.

Monitoring for Protected Areas

There are additional monitoring requirements for protected areas. Protected areas include bodies of surface water and groundwater used for the abstraction of drinking water and habitat and species protection areas identified under the Birds Directive or the Habitats Directive.

In terms of habitat and species protection areas, bodies of water forming these areas must be included in operational monitoring if they are identified (by the Annex II risk assessment and surveillance monitoring) as being at risk of not meeting their environmental objectives. Monitoring must be carried out to assess the magnitude and impact of all relevant significant pressures on these bodies, and where necessary, to assess changes in the status of such bodies resulting from the programmes of measures. Monitoring should also continue until the areas satisfy the water-related requirements of the legislation under which they are designated and have met their objectives under Article 4.

Locations of water bodies to be monitored

It is not economically feasible to monitor all water bodies for all conditions. Therefore, it is necessary to group "similar" water bodies and to select appropriate representative sites for the determination of ecological status for that particular group of sites. While the Directive requires that monitoring be undertaken for all surface and groundwater bodies, grouping is permitted as long as a sufficient number of water bodies are monitored within a group to provide an accurate assessment of status for that group.

Member States should firstly determine which water bodies are required to be monitored in accordance with the Directive. The water bodies selected will vary depending on the objectives of the programme. For example, Annex V of the Directive provides different criteria for the selection of water bodies, depending on whether the objectives of the programme are established to satisfy the requirements for surveillance, operational or investigative monitoring, or for protected areas. Therefore, each Member State must first discriminate according to the specific requirements of the Directive (e.g., size/population boundaries) and eliminate those water bodies in which monitoring is not required.

Once the relevant water bodies have been identified, further grouping may be required due to economic constraints. Water bodies may be grouped based on similar hydrological, geomorphological, geographical or trophic conditions. Alternatively, water bodies could be grouped based on similar catchment impacts or land uses. However, the latter may only be possible in catchments that are dominated by a single land use. Another possibility is to use multivariate classification procedures for identifying groups of sites that form relatively homogeneous areas (although this "black box" approach should be used with caution as there is no guarantee that the composition of the resulting groups will have a recognizable or obvious rationale). Whatever the method by which the water bodies are grouped, it is essential that sufficient water bodies are selected from each group to enable the specific objectives of the monitoring programme to be met with the required levels of precision and confidence.

The characterization required by Annex II makes possible a characterization of water bodies based on environmental variables. Water body characterization as a function of pressures would be made possible through an assessment of pressures and impacts, in which optimization of the monitoring programme could be achieved by a grouping of pressures.

Risk of failing environmental quality objectives

The Directive refers to the identification of water bodies at risk of failing environmental quality objectives as defined in Article 4. This identification will be partially based on existing monitoring data (initially) and then on data arising from surveillance monitoring for subsequent periods of RBMPs. Those water bodies identified as being at risk will then be subject to operational monitoring which will confirm or reject their status in terms of failure to meet the relevant objectives. By implication this means that operational monitoring may need to provide a more precise assessment of the status of those water bodies identified as at risk than that originally obtained from surveillance monitoring.

Not all the Environmental Objectives given in Article 4 will be applicable to all water bodies; they can be summarized as follows:

1. To achieve good groundwater status, good ecological status, good ecological potential or good chemical status;
2. To comply with any standards and objectives associated with Protected Areas;
3. To prevent deterioration in the status of a body of surface water or groundwater;
4. To progressively reduce pollution from priority substances, and cease or phase out emissions, discharges, and losses of priority hazardous substances; and
5. To reverse any significant and sustained upward trend in the concentration of any pollutant in groundwater.

Objectives 1 and 2 imply that assessments will have to be made as to whether the status is better or worse than that which defines the threshold value between good and moderate status (or potential), or is in compliance with defined standards. Objectives 3 to 5 relate to assessing whether the status is deteriorating with time or pollution is decreasing with time. In the latter cases, threshold levels or concentrations of substances against which risk of failure is judged will be specific to the water body of interest and will relate to levels or concentrations specified at a particular time.

As indicated above, the assessment of the risk of failure of a water body will make use (when possible) of data from monitoring stations within the body. The discrimination between good and moderate and hence the risk of failure could be determined based on comparison of the calculated "confidence of compliance" with the appropriate standard or threshold value.

As noted earlier, the assessment of failure would have to consider what would be acceptable Type I and Type II errors. A Type I error would occur when a water body that was truly satisfactory was failed on the evidence of the monitoring programme. Conversely, a Type II error would occur when a water body that was truly unsatisfactory was passed by the monitoring programme.

Transitional Waters

Aspects and features of the different quality elements to be monitored are set out below.

Biological Quality Elements

Phytoplankton

Particularly relevant is the identification of nuisance or potentially toxic species, if they are typical for the transitional water studied. The main difficulties in using phytoplankton as a quality element for transitional waters with pronounced tides are represented by the extremely high natural spatial and temporal variability of the planktonic communities which may make phytoplankton monitoring a useless exercise in some transitional waters. The use of size fraction and size spectra may overcome the problems of taxonomic identification and intercalibration, but it still requires a standardization of methods In shallow environments, the structure of the phytoplankton community can be influenced by the resuspension of benthic microalgae, mostly due to wave and wind.

Seasonal monitoring is suited for representing the phytoplankton community variability when seasonal patterns are predictable. However, the seasonal frequency applies only for taxonomic analyses. At least monthly samplings for phytoplankton chlorophyll *a* should be considered during the vegetation period; weekly sampling would be optimal, fortnightly sampling recommended. Chlorophyll *a* analyses give a coarse assessment of the phytoplankton biomass (expressed as $\mu g\ l^{-1}$), therefore parallel sampling for cell identification and counting should be collected and stored. In case of significant month-by-month changes of chlorophyll *a*, the stored samples might be used for taxonomical analyses. In addition to the chlorophyll *a* analysis, the direct water colour can also give important information, namely the coloured waters are symptoms of typical blooms (e.g., red waters for dinoflagellates, etc.).

Macroalgae (seaweeds)

The main difficulties in using macroalgae as a quality element are represented by the ephemeral behaviour of these quality elements undergoing some spatial and temporal variability which bias monitoring, however, to a much lesser extent than in case of phytoplankton. Therefore in some transitional waters, macroalgae and other macrophytes such as angiosperms may be better suited for monitoring the ecological quality than phytoplankton.

The sampling frequency should be suited for representing changes in seaweed communities, and thus be selected on a region- and type-specific level. During the vegetation period, sampling should be carried out fortnightly to monthly.

Changes in community structure and specific biomasses may be rapid and unpredictable due to the ephemeral characteristics of some of the macroalgae, therefore seasonal samplings are not well suited.

The coverage (as a % of the total system area), changes of this area, the frequency of macroalgal blooms, their size together with the community variability are a good indicator of the state the macroalgae and their environment, and can be used as an early warning system. Qualitative analyses of new species (new forms) can be also performed by site-trained personnel as an additional warning detection.

Angiosperms (seagrasses)

Optional parameters that countries may wish to use in addition are species abundance (as number of individuals per m^2) and biomass (as g dry weight m^{-2}) as well as depth distribution (lower limit of occurrence). Changes in coverage and composition as well as the occurrence of rare or sensitive species may be used as indicators of human, but also natural, impact (e.g., storms, ice winters).

The sampling frequency suited for representing changes in seagrass communities in shallow transitional waters is monthly during the vegetation period. Depending on region and assemblage, it may be sufficient to sample twice during the vegetation period (extensive mapping at a time when species identification is most easy, e.g., during the bloom period, followed by a second survey at the end of the vegetation period).

Benthic invertebrate fauna

Optional parameters that countries may use in addition are biomass (usually expressed as g ash-free dry weight m^{-2}) as well as fractionated biomass (size fractions or body size spectra). However, the reliable determination of macrozoobenthic biomass at a representative station requires a very large number of samples (e.g., 200 replicates per station). Apart from natural small-scale variability, the methodological bias is fairly high due to several steps involved (fresh/wet weight, dry weight, ash-free dry weight). A solution could be to use conversion factors derived from reliable time series taken in the region/type concerned.

A standardization of methods is still required and there is a lack of quality assurance protocols. On a temporal scale, the sampling frequency suited for representing changes in benthic invertebrate communities in shallow transitional waters should be selected on a regional/type-specific basis. Sampling should take place at least twice per year (spring and autumn) A recommendable approach for transitional waters in temperate areas (e.g., river Elbe) is fortnightly sampling during spring/early summer (April–June), followed by 2–3 samplings in August/September. In other areas (e.g., Mediterranean), seasonal sampling might be preferable. Recent attempts to apply statistical analyses to the higher taxonomic levels or on species pooled into ecological or trophic guilds have been successful.

Fish fauna

For classifying the ecological status, the limnological classification scheme based on indicator fish species could be used. Sound abundance estimates require long time series due to high variability. In general, the species composition of transitional waters seems to be most appropriate for WFD purposes; abundance or biomass are not good in these waters because of high variability.

It should be noted that sampling for fish faunal composition and abundance should preferably be carried out at least two times per year (spring/autumn) and that for reliable estimates of fish abundance, long time series of at least ten years are inevitable because of natural variability.

Hydromorphological Quality Elements

An expert's suggestion is to consider the hydrological budget a quality element more general than the freshwater flow, which is actually a component of the hydrological budget. The hydrological budget responds to variation of the freshwater flow but also to variation in the sand accumulation vs. sand erosion processes.

Morphological conditions

Refer to same paragraph in the section on coastal waters.

Depth variations

Refer to same paragraph in the section on coastal waters.

Structure and substrate of the transitional water bed

Refer to same paragraph in the section on coastal waters.

Structure of the transitional zone

The structure of the transitional zone can be monitored in terms of structure of the vegetation occurring at the land-water interfaces, as affected by features of the substrate (mud, sand, rock, etc.), the climatic and hydrologic regimes, and the anthropogenic pressures.

Vegetation coverage, vegetation type, and floristic composition are the parameters that can be monitored.

A major problem is that the structure of vegetation is only an indirect indicator of the activity of the transitional zone as a buffering zone for the pressures of the anthropogenic activities in the watershed.

The structure of vegetation can be monitored every third year.

Hydrological budget

The hydrological budget characterizes the different transitional waters, i.e., estuaries, deltas, lagoons, coastal lakes, ports or gulfs, determines the sediment distribution, and affects the sensitivity and resilience of transitional water ecosystems. Consequently, the hydrological budget has a major influence on all the quality elements in transitional waters.

Hydrological parameters relevant for an estuary are the volumes entering the estuary during high and low tide (tidal volume). The water flow (volume and velocity) is varying very locally. Subsequently, erosion and sedimentation processes are sensitive to anthropogenic measures (LT-process) and extreme events like storms (ST-process). Special attention has to be given to the fish-breeding areas between 0 to 5 m water depth and currents below 0.5 m. Monitoring these areas should be included in the programme.

Changes in the components of the hydrological budget, due to human activities, are expected to be relatively slow. Therefore, monitoring is recommended every three years. Monitoring should be performed with data collection on all the freshwater inputs and outputs arranged on a seasonal scale.

Chemical and Physico-chemical Quality Elements

For the chemical and physico-chemical quality elements, refer to the same paragraphs under coastal waters.

<u>A specific consideration for transitional waters is:</u>

Salinity

It is fundamental to measure the salinity gradient horizontally as well as vertically, especially for the physical delimitation of the transitional zone.

Coastal Waters

Biological Quality Elements

A very important issue when using biological elements as QE is the need of expertise required for taxonomic identification at the species level and the *in-situ* taxonomic resolution limitation.

Appropriately scientifically qualified personnel are to carry out the surveys. They shall be able to document competence within their specialist field, and participate in ring-testing, when the appropriate routines are available. For investigations spanning several years, priority should be given to continuity in personnel carrying out the recordings.

Phytoplankton

Particularly relevant is the identification of nuisance or potentially toxic species as important assessment parameters. Bloom frequency and intensity is considered an indicative parameter for classification of ecological status.

High natural spatial and temporal variability of the planktonic communities requires frequent sampling to ensure meaningful data for classification or detection of events (blooms). Sampling frequency is determined by the variability, and a minimum of monthly sampling is recommended with optional increased sampling frequency in seasons with main bloom events. Sampling should be performed together with measurements of chemical and physico-chemical parameters. Seasonal sampling is a minimum frequency.

The minimum sampling frequency required by the Directive is every six months. However, available expert knowledge and pilot studies on sampling frequencies could be helpful to set up the most appropriate sampling frequency, number and location of stations on a regional or type-specific level. A selection of region/area-specific phytoplankton indicator species could be useful.

New monitoring programmes for the WFD could build on the existing phytoplankton monitoring programmes for other purposes, e.g., the Shellfish Hygiene Directive (Council Directive 91/492/EEC of 15 July 1991), to ensure best "value for money" in monitoring.

Macroalgae/Angiosperms (Phytobenthos)

It is important to monitor not only their composition and abundance (as requested in the Directive), but also their distributions, extension, and variation in time and space (mapping at different needed scales), as this provides important information not only on the health status of the plants' habitats, but also on the ecosystem stability, as variations may indicate long-term changes in the physical conditions at the site.

Macroalgae are an important region-specific parameter. Macroalgal communities often include a wide range of species/functional groups that may change upon eutrophication, e.g., highly diverse algal species can be replaced by opportunistic and stress-resistant seaweeds.

For angiosperms, distribution is the most important parameter because changes are not occurring from month to month. It may therefore be sufficient to monitor angiosperms every six months (spring/autumn), once a year or even only once every three years, depending on the species.

Supplementary variables essential for interpretation of macrophytobenthos results include: substrate type, depth in relation to sea level or standard datum, slope and bearing, presence of loose sediment, degree of wave exposure, tidal range, Secchi disk depth, and salinity.

Benthic invertebrate fauna

The required parameters to be measured are composition and abundance. Important variables to be considered are also diversity of species and presence of sensitive or higher taxa as well as biomass, the latter being indicative of eutrophication phenomena.

Recent studies in taxonomic classification have shown that looping species into higher taxa (including morphological categories) does not necessarily limit the sensitivity of animal assemblages to detect impacts.

It should be noted that sometimes it is difficult to show a clear correlation between possible changes found in the benthos (e.g., long-term changes in zoobenthos species composition) and eutrophication. Biomass may be a better parameter, though not mandatory for WFD monitoring. Therefore, it is recommended to include biomass as an optional monitoring parameter. Furthermore, it should be noted that other factors, e.g. fisheries, may have an over-riding effect compared with eutrophication effects. A distinction should be made between acute, direct effects on the benthos (e.g., directly related to dredging or oxygen deficiency and/or toxic blooms) and long-term changes. Both may need different sampling frequencies and spatial coverage.

Hydromorphological Quality Elements

Morphological conditions

The morphological characteristics of coastal areas are generally subject to low variability due to natural large-scale bottom dynamics processes or changes in tidal regime and weather patterns.

Relevant for ecological status is the time scale of the changes resulting from human impact in the past. A time scale of 10–25 years means that relevant changes in hydromorphological conditions have an impact on ecology. In addition, sea level rise makes it necessary to adapt the monitoring frequency and spatial scale to analyse the processes and to find the sand budgets in coastal zones, sheltered seas, and estuaries.

Monitoring the trends in depth gradients has to take into account water management measures like dredging and dumping activities and naturally induced variability, under particular weather conditions such as storm events and ice winters/ice coverage. It must also consider natural coastal erosion and elevation of the land, e.g., in the Baltic Sea.

Depth variations

The topography of the area (shape, bathymetry, slope) influences the biological communities living in it. Depth variations could be important elements to be monitored in areas where disturbances are expected. Anthropogenic changes will have relevance for the status classification of the water body.

Structure and substrate of the coastal bed

Changes in morphological conditions and/or the nature of the substratum may exert severe detrimental effects on benthic organisms. Differences between communities in coastal zones and estuaries are linked to a coastal typology:

Possible causes of anthropogenic alterations in structure, substrate, and shape of the coastal bed are:

- coastal constructions (dredging, dumping, dams, artificial reefs, etc.);
- variations in riverine sediment inputs (solid transport regime) due to human impact.

For depth variation and structure and substrate of the coastal bed, it may be sufficient to collect the required information once (e.g., a map of the coastal bed) and to record:

- at each sampling carried out after the first thorough survey: typical parameters (e.g., nature of substratum) and obvious changes (e.g., visible changes after big storm events);
- changes due to anthropogenic impact (e.g., dam construction).

A thorough survey should be repeated in regular, but longer intervals (e.g., once per management period or longer, depending on parameter).

Structure of the intertidal zone

As for the structure of the intertidal zone, it cannot be used as a quality element in the Mediterranean or the Baltic eco-regions, given the low amplitude of tides in the Mediterranean basin and in the Baltic Sea.

Thus it has been proposed to introduce the intertidal/***mediolittoral*** term as its ecological relevance is due to the fact that it comprises living assemblages that require or tolerate immersion but cannot survive permanent or semi-permanent immersion (same definition for the intertidal). Thus, the mediolittoral zone supports diverse and very productive assemblages of algae and invertebrates that can be considered analogues to those of intertidal habitats.

Possible causes of anthropogenic alterations in structure, substrate, and shape of the intertidal zone are:

- coastal constructions (dredging, dumping, dams, artificial reefs, etc.);
- chemical inputs (nutrients) leading to a change in the composition of macroalgal communities;
- variations in coastal or riverine sediment movements (sediment transport regime) due to human impact.

Mediterranean experts' judgements suggest to focus particular attention on the structure and condition of the mediolittoral and upper infralittoral zones in tideless seas, at least in the Mediterranean, since several species and communities thriving in this area are very good biological indicators, as exposed to a wide range of anthropogenic impacts due to their critical position at the interface between the sea and the land.

Tidal regime

The tidal regime in terms of the direction of dominant currents and level of wave exposure can be seasonally predictable and is available from most of the National Hydrographic Services. Deviations from the natural pattern in tidal regime derive from direct anthropogenic intervention on the profile of the coastline and may have severe bearings on the stability of the biological assemblages, thus they need to be taken into consideration. Asymmetry in the tidal waves results in positive or negative yearly budgets of sediments.

Due to the low tidal range in the Mediterranean and Baltic Seas, tidal currents play a very minor role, if any. It is the case also in part of the North Sea, e.g., the Skagerrak.

Direction of dominant currents

The direction and intensity (speed) of currents represent the main hydromorphological quality elements influencing the biological elements. They could be important elements to be monitored in areas where anthropogenic disturbance could be relevant for the status classification of the water body.

These parameters assume quite a relevant importance in those eco-regions and specific areas where the tidal range is very low and thus poorly influences the coastal processes.

Mainly changes in hydrodynamics induced by morphological changes will result in relevant ecological effects. Temporal changes (storms, anthropogenic activities) could be balanced on a time scale of 5–6 years. On local scales, this could not be the case. Monitoring should take into account these short term-effects.

Wave exposure

Wave exposure (wave height, wind, Fetch-index) varies considerably according to coastal typology (from highly exposed to very sheltered) and meteorological conditions, in the different eco-regions. Parameters to be monitored in case of anthropogenic disturbances are, e.g., frequencies of storms, directions, and high/low tide surge levels.

Chemical and Physico-chemical Quality Elements

In most of the EC countries, all these parameters (with the exception of specific pollutants) are routinely measured as part of their national monitoring programmes, with a variable frequency (weekly to monthly), using national guidelines or OSPAR/HELCOM standards.

Transparency

Transparency is mainly affected by mineral turbidity, organic pollution (e.g., urban discharges), and eutrophication; it can naturally vary due to local hydrodynamics, river discharge, and seasonal plankton blooms.

The transparency parameter is necessary for the determination of the depth of the euphotic layer, where primary production exceeds respiration. Measurement is difficult in "troubled waters", e.g., the Northeast Atlantic Wadden Sea with high loads of resuspended sediments.

Thermal conditions

Temperature profiles along the water column can be easily obtained by means of *in situ* autographic instruments. The thermal structure of the water column is relevant information for assessing mixing/stratification conditions, which strongly influence primary production as well as possibly the development of oxygen deficiency.

Oxygen conditions

Dissolved oxygen concentration is subject to high natural variability since its solubility depends on temperature and salinity. Deviation, in absolute value, of % saturation from 100% is indicative of intense primary production and/or organic pollution.

Salinity

Salinity in coastal waters can be subject to high natural variability due to freshwater inputs and mixing of water masses, and due to tidal currents.

Salinity measurements in coastal waters can be used to detect freshwater ingressions from the land; the dilution rate of nearshore waters varies considerably in different areas and can be used, together with other quality elements, to indicate potential pollution.

Nutrient conditions

The concentrations of nutrients, together with the concentration of chlorophyll *a*, provide information on the general trophic conditions.

Natural variability of nutrient concentrations can be relevant on a seasonal basis; in coastal waters, high nutrient concentrations, mainly related to riverine inputs, are indicative of eutrophication and/or organic pollution.

In order to enable discrimination of pollution sources, the following parameters should be analysed:

- Total Phosphorus (TP, $\mu g\ l^{-1}$)
- Soluble Reactive Orthophosphate ($P\text{-}PO_4$, $\mu g\ l^{-1}$)
- Total Nitrogen (TN, $\mu g\ l^{-1}$)
- Nitrate+Nitrite ($N\text{-}NO_3 + N\text{-}NO_2$, $\mu g\ l^{-1}$)
- Ammonia ($N\text{-}NH_4$, $\mu g\ l^{-1}$)
- An additional parameter is silicate ($Si\text{-}SiO_3$, $\mu g\ l^{-1}$), which is a growth requirement for diatoms.
- For a better understanding of nutrient cycling in coastal waters, the following supplementary parameters are recommended:
- Particulate Organic Carbon (POC-C, $\mu g\ l^{-1}$)
- Particulate Organic Nitrogen (PON-N, $\mu g\ l^{-1}$)
- Particulate Organic Phosphorus (POP-P, $\mu g\ l^{-1}$)

Nutrient ratios (N/P/Si) are useful for the interpretation of results and eutrophication status.

Discussion and implications of WFD for mariculture

Definition of water bodies

At the present time, national competent authorities are in the process of applying the CIS guidelines to dividing River Basin Districts into water bodies.

According to the Directive, water bodies are to be defined on a range of physico-chemical factors including location, tidal range, salinity, current velocity, exposure, temperature, mixing characteristics, etc. These factors will probably lead authorities to define quite large water bodies, probably on the scale of kilometres to low tens of kilometres. It is therefore likely that mariculture sites would be considered to be located within larger water bodies, and to act as pressures on the water quality within the wider water body.

However, the definition of water bodies has to take account of the particular pressures that may impact the ecological quality of surface waters in the area. Pollution control authorities will use the water body as the primary unit for pollution assessment, control and regulation in the future. From that point of view, arguments can be expressed calling for a one-to-one relationship between water bodies and potentially significant sources of pollution. Such arguments could lead to the definition of rather small water bodies, perhaps on the scale of single areas leased for mariculture.

The consequences for mariculture of the debate between the above two points of view are considerable. In the first case, in which larger water bodies are defined, the localized impact of mariculture can be viewed in the context of the wider environment of the water body. It would then be appropriate to assess the pressure from mariculture on the overall ecological quality of the water body.

In the second case, attention is closely focused on the effect of the pressures on the ecological quality immediately surrounding the mariculture unit. Such close focus would increase the likelihood of these small water bodies failing to meet the target of good ecological quality. For example, the well-established localized enrichment of seabed sediments arising from fish farming (and to a lesser degree shellfish cultivation) is known to commonly result in alterations to the benthic infaunal community.

It would appear that the former approach is more consistent with the general philosophy underlying the WFD. The debate is not confined to pressures arising from mariculture, but arises in the same way in relation to other activities operating within, and impacting on, the coastal zone, such as domestic waste disposal, industrial effluent discharge, farming, forestry, etc.

It is clear that there are still significant uncertainties in the definition of water bodies and the subsequent classification process. It is not clear, however, what approach national authorities will take to the definition of water bodies, although this should become clearer over the next year as authorities progress through the assessment of pressures on water bodies and the consequential impacts.

Classification

The initial purpose of monitoring under WFD (after the definition of reference conditions) is to provide information on which to base the classification of water bodies. Monitoring in transitional and/or coastal waters will include biological, hydromorphological, and physico-chemical elements.

The WFD contains definitions of the status of water bodies, based upon assessments under each of the elements of the monitoring programme. By way of example, these are reproduced below for transitional waters. It should be noted that the target in the Directive is that all water bodies should attain good status by 2015.

Phytoplankton

High status	Good status	Moderate Status
The composition and abundance of the phytoplanktonic taxa are consistent with undisturbed conditions. The average phytoplankton biomass is consistent with the type-specific physico-chemical conditions and is not such as to significantly alter the type-specific transparency conditions. Planktonic blooms occur at a frequency and intensity which is consistent with the type-specific physico-chemical conditions.	There are slight changes in the composition and abundance of phytoplanktonic taxa. There are slight changes in biomass compared to the type-specific conditions. Such changes do not indicate any accelerated growth of algae resulting in undesirable disturbance to the balance of organisms present in the water body or to the physico-chemical quality of the water. A slight increase in the frequency and intensity of the type-specific planktonic blooms may occur.	The composition and abundance of phytoplanktonic taxa differ moderately from type-specific conditions. Biomass is moderately disturbed and may be such as to produce a significant undesirable disturbance in the condition of other biological quality elements. A moderate increase in the frequency and intensity of planktonic blooms may occur. Persistent blooms may occur during summer months.

Macroalgae

High status	Good status	Moderate status
The composition of macroalgal taxa is consistent with undisturbed conditions. There are no detectable changes in macroalgal cover due to anthropogenic activities.	There are slight changes in the composition and abundance of macroalgal taxa compared to the type-specific communities. Such changes do not indicate any accelerated growth of phytobenthos or higher forms of plant life resulting in undesirable disturbance to the balance of organisms present in the water body or to the physico-chemical quality of the water.	The composition of macroalgal taxa differs moderately from type-specific conditions and is significantly more distorted than at good quality. Moderate changes in the average macroalgal abundance are evident and may be such as to result in an undesirable disturbance to the balance of organisms present in the water body.

Angiosperms

High status	Good status	Moderate status
The taxonomic composition corresponds totally or nearly totally to undisturbed conditions. There are no detectable changes in angiosperm abundance due to anthropogenic activities.	There are slight changes in the composition of angiosperm taxa compared to the type-specific communities. Angiosperm abundance shows slight signs of disturbance.	The composition of the angiosperm taxa differs moderately from the type-specific communities and is significantly more distorted than at good quality. There are moderate distortions in the abundance of angiosperm taxa.

Benthic invertebrate fauna

High status	Good status	Moderate status
The level of diversity and abundance of invertebrate taxa is within the range normally associated with undisturbed conditions. All the disturbance-sensitive taxa associated with undisturbed conditions are present.	The level of diversity and abundance of invertebrate taxa is slightly outside the range associated with the type-specific conditions. Most of the sensitive taxa of the type-specific communities are present.	The level of diversity and abundance of invertebrate taxa is moderately outside the range associated with the type-specific conditions. Taxa indicative of pollution are present. Many of the sensitive taxa of the type-specific communities are absent.

Fish

High status	Good status	Moderate status
Species composition and abundance are consistent with undisturbed conditions.	The abundance of the disturbance-sensitive species shows slight signs of distortion from type-specific conditions attributable to anthropogenic impacts on physico-chemical or hydromorphological quality elements.	A moderate proportion of the type-specific disturbance-sensitive species are absent as a result of anthropogenic impacts on physico-chemical or hydromorphological quality elements.

Hydromorphological elements

High status	Good status	Moderate status
Morphological conditions:		
Depth variations, substrate conditions, and both the structure and condition of the inter-tidal zones correspond totally or nearly totally to undisturbed conditions.	Conditions consistent with the achievement of the values specified above for the biological quality elements.	Conditions consistent with the achievement of the values specified above for the biological quality elements.
Tidal regime:		
The freshwater flow regime corresponds totally or nearly totally to undisturbed conditions.	Conditions consistent with the achievement of the values specified above for the biological quality elements.	Conditions consistent with the achievement of the values specified above for the biological quality elements.

Physico-chemical elements

High status	Good status	Moderate status
General conditions:		
The physico-chemical elements correspond totally or nearly totally to undisturbed conditions. Nutrient concentrations remain within the range normally associated with undisturbed conditions. Temperature, oxygen balance, and transparency do not show signs of anthropogenic disturbance and remain within the ranges normally associated with undisturbed conditions.	Temperature, oxygen conditions, and transparency do not reach levels outside the ranges established so as to ensure the functioning of the ecosystem and the achievement of the values specified above for the biological quality elements. Nutrient concentrations do not exceed the levels established so as to ensure the functioning of the ecosystem and the achievement of the values specified above for the biological quality elements.	Conditions consistent with the achievement of the values specified above for the biological quality elements.
Specific synthetic pollutants:		
Concentrations close to zero and at least below the limits of detection of the most advanced analytical techniques in general use.	Concentrations not in excess of the standards set in accordance with the procedure detailed above without prejudice to Directive 91/414/EC and Directive 98/8/EC. (<environmental quality standard).	Conditions consistent with the achievement of the values specified above for the biological quality elements.
Specific non-synthetic pollutants:		
Concentrations remain within the range normally associated with undisturbed conditions (background levels).	Concentrations not in excess of the standards set in accordance with the procedure detailed above without prejudice to Directive 91/414/EC and Directive 98/8/EC. (<environmental quality standard).	Conditions consistent with the achievement of the values specified above for the biological quality elements.

Table A9.4 sets out the various quality elements to be considered in the Water Framework Directive and the predicted degree of interaction with mariculture activities. It is possible to make an assessment of these elements with respect to their potential to be affected by mariculture activities (see Table A9.4). In most cases where impact is likely (macroalgae, benthic invertebrates, nature of substrate, etc.), the effects are also predicted to be rather local, in the area immediately surrounding the farm.

In only a few cases there may be some potential for more widespread effects (e.g., increased phytoplankton growth as a consequence of nutrient concentrations, or toxic effects from specific pollutants). However, in some cases, the degree of impact may be quite strong; for example, the benthic invertebrate infauna below fish cages may be strongly modified compared to conditions immediately outside the area of deposition of waste from the cages.

The purpose of the classification process is to assess the ecological quality of each water body against the reference condition defined for that particular type. The reference condition is a description of the biological quality elements that exist, or would exist, at high status, that is, with no, or very minor, disturbance from human activities. Reference condition standards are required to enable the assessment of ecological quality against these standards.

Table A9.4. Quality elements to be considered in the Water Framework Directive and their relation to mariculture activities.

Elements of monitoring in Transitional and Coastal Waters	Predicted degree of interaction with mariculture activities
Biological elements	
• *Composition, abundance, and biomass of phytoplankton*	• Unlikely except in areas of low water exchange and high exploitation
• *Composition and abundance of other aquatic flora*	• Possible local impact from nutrient release
• *Composition and abundance of benthic invertebrate fauna*	• Likely local impact
• *Composition and abundance of fish fauna*	• Unlikely
Hydromorphological elements supporting the biological elements:	
Morphological conditions:	
• *depth variation*	• Unlikely to be significant in most situations
• *quantity, structure, and substrate of the bed*	• Local impacts are common
• *structure of the inter-tidal zone*	• Unlikely, except in intertidal areas used for mollusc cultivation (e.g., oyster trestles, mussel bouchots)
Tidal regime:	
• *freshwater flow*	• Nil
• *wave exposure*	• Local, and probably rarely significant
Chemical and physio-chemical elements supporting the biological elements:	
General:	
• *Transparency*	• Possible local impact
• *Thermal conditions*	• Nil
• *Salinity*	• Nil
• *Oxygen conditions*	• Possible local impact
• *Nutrient conditions*	• Local or wider impacts on concentrations are likely
Specific Pollutants:	
• *Pollution by all priority substances identified as being discharged into the body of water*	• Probably insignificant impact from priority hazardous substances
• *Pollution of other substances identified as being discharged in significant quantities into the body of water.*	• Site-specific pressure and risk assessments required for some substances such as antifoulants and sea lice treatments

It is clear that reference conditions for water body types will need to be framed in such a way that the natural variability present in the environment at high or good status can be accommodated. Water body types will contain a sufficient range of environments (for example, differences in substrate, degree of exposure, etc.) that would lead to rather different communities of animals or plants being present. There is currently considerable work in progress to develop a coherent system of reference conditions for the water body types being proposed for transitional and marine waters.

In addition to the variability arising from natural factors, water bodies will also include some additional variability arising from anthropogenic factors, such as coastal works or waste discharges. As indicated above, the definitions of good ecological quality contain rather qualitative statements. For example, the description of good status for benthic fauna in transitional waters is described as:

"The level of diversity and abundance of invertebrate taxa is slightly outside the range associated with the type-specific conditions.

Most of the sensitive taxa of the type-specific communities are present."

National authorities are currently developing classification schemes based upon the principles defined in the Directive. The operation of the various schemes, and the conclusions to be drawn in terms of classification, will be the subject of intercomparison exercises, primarily within eco-regions (as defined in the WFD).

It is not yet clear how the national schemes, and subsequently the intercompared schemes, will accommodate differences in the values of biological or hydrochemical elements within water bodies. How this is to be done is clearly of importance to mariculture activities, as mariculture sites will present pressures on the environment and some of the elements of the assessment will be at less than reference status at these sites. Again, this question is not confined to mariculture. Many other anthropogenic activities that result in waste discharges are subject to the same uncertainties.

The development of typology and classification systems for transitional and coastal waters, as well as the criteria for the assessment of water quality status for each water body type, is likely to lead to a more harmonized and internationally comparable approach to monitoring.

Measures to improve ecological quality (mitigation measures)

The overall aim of the Water Framework Directive is the achievement of good water status in all waters by 2015. It is probable that the initial classification will result in some water bodies being classified as having an ecological status below the target level. In such cases, Member States will then be required to take steps to improve the status of these water bodies.

"Member States should adopt measures to eliminate pollution of surface waters by the priority substances and progressively to reduce pollution by other substances which would otherwise prevent Member States from achieving the objectives for the bodies of surface water."

"...... specific measures for the progressive reduction of discharges, emissions and losses of priority substances ...."

At present, it is not clear what measures/actions Member States may choose to take. It is apparent that rather little consideration has yet been given to this aspect of the Directive, with most attention being given to prior activities (such as the development of reference conditions, monitoring, and subsequent classification) with more pressing deadlines. However, it can be anticipated that additional management and mitigative actions may be required of aquaculture operations in some areas where good ecological status has not been achieved.

It is clear that the promotion of sustainable European aquaculture does not conflict with the requirements of the WFD. Aquaculture should therefore not seek "special treatment". Rather, aquaculture is to be viewed as a coastal zone activity in the same manner as other activities that can influence, and have requirements for, good coastal water quality, such as domestic waste disposal, agriculture, tourism, and forestry. The contribution of aquaculture to ecological conditions assessed as less than good should be considered in the same manner as the contributions from other activities.

In developing mitigation schemes, however, it should be noted that aquaculture can also contribute to improving ecological status of surface water bodies. Such activities could include:

- macroalgal cultivation which can remove significant amounts of nutrients from the surrounding waters;
- bivalve mollusc cultivation which can extract both nutrients and contaminants from the water column through their filtering activity;
- polyculture, e.g., finfish and macroalgae in which the inorganic nutrients from the finfish farms are taken up by the macroalgae and in which both products can have an economic value.

In addition to the above, other actions could be taken by the aquaculture industry that could lead to a reduction and minimization of emissions of "specific pollutants" to the environment and a minimization of impacts. These may include:

- improved feed formulations that could lead to an improved food conversion ratio (FCR) and hence a reduction in feed loss to the environment;
- improvement in feeding methods that could lead to a reduction in feed loss to the environment;
- improved sea-lice control strategies that could lead to a reduction in the use of anti-lice chemical treatments and minimization in emissions to the receiving environment;
- improvement in cage technology that could allow the siting of fish farms in more exposed and better flushed areas, thus reducing the impact on both the sediments and receiving waters;
- development of artificial reefs in conjunction with fish farms which could enhance the development of an epifauna filter-feeding community to act as biofilter and enlarge and allow rehabilitation of stressed natural habitats, and potentially quantitatively boost many marine species;
- development of new methods for under-cage collection of wastes.

Conclusions

It is clear that the implementation of the Water Framework Directive, within the climate created by the Commission policy for Sustainable Aquaculture, has the potential to have significant impacts on mariculture activities as they are currently undertaken. The above discussion has identified the importance of national (and international?) policies adopted on the definition of water bodies, the definition of reference conditions, and the interpretation of monitoring data. These will influence the likelihood of mariculturists being required to conform with any additional measures that may be instigated by Member States to ensure the achievement of high status in all surface water bodies. Mariculture also offers opportunities to regulators who wish to reduce the impacts of discharges from other activities.

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ANNEX 10: ICES GUIDELINES FOR THE MANAGEMENT OF MARINE SEDIMENT EXTRACTION[1]

Introduction

In many countries, sand and gravel[2] dredged from the seabed makes an important contribution to the national demand for aggregates, directly replacing materials extracted from land-based sources. This reduces the pressure to work land of agricultural importance or environmental and hydrological value, and where materials can be landed close to the point of use, there can be additional benefits of avoiding long distance over-land transport. Marine dredged sand and gravel is also increasingly used in flood and coastal defence, fill and land reclamation schemes. For beach replenishment, marine materials are usually preferred from an amenity point of view, and are generally considered to be the most appropriate economically, technically, and environmentally.

However, these benefits need to be balanced against the potential negative impacts of aggregate dredging. Aggregate dredging activity, if not carefully controlled, can cause significant damage to the seabed and its associated biota, to commercial fisheries and to the adjacent coastlines, as well as creating conflict with other users of the sea. In addition, current knowledge of the resource indicates that while there are extensive supplies of some types of marine sand, there appear to be more limited resources of gravel suitable, for example, to meet current concrete specifications and for beach nourishment.

Against the background of utilizing a finite resource, with the associated environmental impacts, it is recommended that regulators develop and work within a strategic framework which provides a system for examining and reconciling the conflicting claims on land and at sea. Decisions on individual applications can then be made within the context of the strategic framework.

General principles for the sustainable management of all mineral resources overall include:

- conserving minerals as far as possible, whilst ensuring that there are adequate supplies to meet the demands of society;
- encouraging their efficient use (and, where appropriate, re-use), minimizing wastage and avoiding the use of higher quality materials where lower grade materials would suffice;
- ensuring that methods of extraction minimize the adverse effects on the environment, and preserve the overall quality of the environment once extraction has ceased;
- the encouragement of an ecosystem approach to the management of extraction activities and identification of areas suitable for extraction;
- protecting sensitive areas and important habitats (such as marine conservation areas) and industries (including fisheries) and the interests of other legitimate uses of the sea;
- preventing unnecessary sterilization of mineral resources by other forms of development.

The implementation of these principles requires a knowledge of the resource, and an understanding of the potential impacts of its extraction and of the extent to which rehabilitation of the seabed is likely to take place. The production of an Environmental Statement, developed along the lines suggested below, should provide a basis for determining the potential effects and identifying possible mitigating measures. There will be cases where the environment is too sensitive to disturbance to justify the extraction of aggregate, and unless the environmental and coastal issues can be satisfactorily resolved, extraction should not normally be allowed.

It should also be recognized that improvements in technology may enable exploitation of marine sediments from areas of the seabed which are not currently commercially viable, while development of technical specifications for concrete, etc., may in the future enable lower quality materials to be used for a wider range of applications. In the shorter term, continuation of programmes of resource mapping may also identify additional sources of coarser aggregates.

Scope

It is recognized that sand and gravel extraction, if undertaken in an inappropriate way, may cause significant harm to the marine and coastal environment. There are a number of international and regional initiatives that should be taken into account when developing national frameworks and guidelines. These include the Convention on Biological Diversity (CBD), EU Directives (particularly those on birds, habitats, Environmental Impact Assessment (EIA), and Strategic Environmental Assessment (SEA)—once implemented), and other regional conventions/agreements, in particular the OSPAR and Helsinki Conventions, and initiatives pursued under them. This subject, for example, has recently been included in the Action Plan for Annex V to the 1992 OSPAR Convention, on the Protection and Conservation of the Ecosystems and Biological Diversity of the Maritime Area, as a human activity requiring

[1] These guidelines do not relate to navigational dredging (i.e., maintenance or capital dredging).

[2] It is recognized that other materials are also extracted from the seabed, such as stone, shell, and maerl, and similar considerations should apply to them.

assessment. It is also recognized that certain ecologically sensitive areas may not be designated under international, European, or national rules but nonetheless require particular consideration within the assessment procedures described in these Guidelines.

Administrative framework

It is recommended that countries have an appropriate framework for the management of sand and gravel extraction and that they define and implement their own administrative framework with due regard to these guidelines. There should be a designated regulatory authority to:

- issue authorization having fully considered the potential environmental effects;
- be responsible for compliance monitoring;
- develop the framework for monitoring;
- enforce conditions.

Environmental impact assessment

The extraction of sand and gravel from the seabed can have significant physical and biological effects on the marine and coastal environment. The significance and extent of the environmental effects will depend upon a range of factors including the location of the extraction area, the nature of the surface and underlying sediment, coastal processes, the design, method, rate, amount and intensity of extraction, and the sensitivity of habitats and assorted biodiversity, fisheries, and other uses in the locality. These factors are considered in more detail below. Particular consideration should be given to sites designated under international, European, national and local legislation, in order to avoid unacceptable disturbance or deterioration of these areas for the habitats, species, and other designated features.

To enable the organization(s) responsible for authorizing extraction to evaluate the nature and scale of the effects and to decide whether a proposal can proceed, it is necessary that an adequate assessment of the environmental effects be carried out. It is important, for example, to determine whether the application is likely to have an effect on the coastline, or have potential impact on fisheries and the marine environment.

The Baltic Marine Environment Protection Commission (Helsinki Commission) adopted HELCOM Recommendation 19/1 on 26 March 1998. This recommends to the Governments of Contracting Parties that an EIA should be undertaken in all cases before an extraction is authorized. For EU Member States, the extraction of minerals from the seabed falls within Annex II of the "Directive on the Assessment of the Effects of Certain Public and Private Projects on the Environment" (85/337/EEC). As an Annex II activity, an EIA is required if the Member State takes the view that one is necessary. It is at the discretion of the individual Member States to define the criteria and/or threshold values that need to be met to require an EIA. The Directive was amended in March 1997 by Directive 97/11/EC. Member States are obliged to transpose the requirements of the Directive into national legislation by March 1999.

It is recommended that the approach adopted within the EU be followed. Member States should therefore set their own thresholds for deciding whether and when an EIA is required, but it is recommended that an EIA always be undertaken where extraction is proposed in areas designated under international, European, or national rules and in other ecologically sensitive areas. For NATURA 2000 sites, Article 6 of the Habitats Directive contains special requirements in this respect.

Where an EIA is considered appropriate, the level of detail required to identify the potential impacts on the environment should be carefully considered and identified on a site-specific basis. An EIA should normally be prepared for each extraction area, but in cases where multiple operations in the same area are proposed, a single impact assessment for the whole area may be more appropriate, which takes account of the potential for any cumulative impacts. In such cases, consideration should be given to the need for a strategic environmental assessment.

Consultation is central to the EIA process. The framework for the content of the EIA should be established by early consultation with the regulatory authority, statutory consultees, and other interested parties. Where there are potential transboundary issues, it will be important to undertake consultation with the other countries likely to be affected, and the relevant Competent Authorities are encouraged to establish procedures for effective communication.

As a general guide, it is likely that the following topics considered below will need to be addressed.

Description of the physical setting

The proposed extraction area should be identified by geographical location, and described in terms of:

- the bathymetry and topography of the general area;
- the distance from the nearest coastlines;
- the geological history of the deposit;
- the source of the material;
- the type of material;
- sediment particle size distribution;
- the extent and volume of the deposit;
- the stability and/or natural mobility of the deposit;

- the thickness of the deposit and evenness over the proposed extraction area;
- the nature of the underlying deposit, and any overburden;
- local hydrography including tidal and residual water movements;
- wind and wave characteristics;
- the average number of storm days per year;
- an estimate of bed-load sediment transport (quantity, grain size, direction);
- the topography of the seabed, including occurrence of bedforms;
- the existence of contaminated sediments and their chemical characteristics;
- natural (background) suspended sediment load under both tidal currents and wave action.

Description of the biological setting

The biological setting of the proposed extraction site and adjacent areas should be described in terms of:

- the flora and fauna within the area likely to be affected by aggregate dredging (e.g., pelagic and benthic community structure), taking into account temporal and spatial variability;
- information on the fishery and shellfishery resources including spawning areas, with particular regard to benthic spawning fish, nursery areas, over-wintering grounds for ovigerous crustaceans, and known routes of migration;
- trophic relationships (e.g., between the benthos and demersal fish populations by stomach content investigations);
- the presence of any areas of special scientific or biological interest in or adjacent to the proposed extraction area, such as sites designated under local, national or international regulations (e.g., Ramsar sites, the UNEP "Man and the Biosphere" Reserves, World Heritage sites, Marine Protected Areas (MPAs), Marine Nature Reserves, Special Protection Areas (Birds Directive), or the Special Areas of Conservation (Habitats Directive)).

Description of the proposed aggregate dredging activity

The assessment should include, where appropriate, information on:

- the total volume to be extracted;
- proposed maximum annual extraction rates and dredging intensity;
- the expected lifetime of the resource and proposed duration of aggregate dredging;
- the aggregate dredging equipment to be used;
- spatial design and configuration of aggregate dredging (i.e., the maximum depth of deposit removal, the shape and area of resulting depression);
- substrate composition on cessation of aggregate dredging;
- proposals to phase (zone) operations;
- whether on-board screening (i.e., rejection of fine or coarse fractions) will be carried out;
- the number of dredgers operating at a time;
- routes to be taken by aggregate dredgers to and from the proposed extraction area;
- the time required for aggregate dredgers to complete loading;
- the number of days per year on which aggregate dredging will occur;
- whether aggregate dredging will be restricted to particular times of the year or parts of the tidal cycle;
- the direction of aggregate dredging (e.g., with or across tide).

It may be appropriate, when known, also to include details of the following:

- energy consumption and gaseous emissions;
- ports for landing materials;
- servicing ports;
- on-shore processing and onward movement;
- project-related employment.

Information required for physical impact assessment

To assess the physical impacts, the following should be considered:

- implications of extraction for coastal and offshore processes, including possible effects on beach draw down, changes to sediment supply and transport pathways, changes to wave and tidal climate;
- changes to the seabed topography and sediment type;
- exposure of different substrates;
- changes to the behaviour of bedforms within the extraction and adjacent areas;
- the potential risk of release of contaminants by aggregate dredging, and exposure of potentially toxic natural substances;

- the transport and settlement of fine sediment disturbed by the aggregate dredging equipment on the seabed, and from hopper overflow or on-board processing and its impact on normal and maximum suspended load;
- the effects on water quality mainly through increases in the amount of fine material in suspension;
- implications for local water circulation resulting from removal or creation of topographic features on the seabed;
- the time scale for potential physical "recovery" of the seabed.

Information required for biological impact assessment

To assess the biological impact, the following information should be considered:

- changes to the benthic community structure, and to any ecologically sensitive species or habitats that may be particularly vulnerable to extraction operations;
- effects of aggregate dredging on pelagic biota;
- effects on the fishery and shellfishery resources including spawning areas, with particular regard to benthic spawning fish, nursery areas, over-wintering grounds for ovigerous crustaceans, and known routes of migration;
- effects on trophic relationships (e.g., between the benthos and demersal fish populations);
- effects on sites designated under local, national or international regulations (see above);
- the predicted rate and mode of recolonization, taking into account initial community structure, natural temporal changes, local hydrodynamics, and any predicted change of sediment type;
- effects on marine flora and fauna including seabirds and mammals;
- effects on the ecology of boulder fields/stone reefs.

Interference with other legitimate uses of the sea

The assessment should consider the following in relation to the proposed programme of extraction:

- commercial fisheries;
- shipping and navigation lanes;
- military exclusion zones;
- offshore oil and gas activities;
- engineering uses of the seabed (e.g., adjacent extraction activities, undersea cables and pipelines including associated safety and exclusion zones);
- areas designated for the disposal of dredged or other materials;
- location in relation to existing or proposed aggregate extraction areas;
- location of wrecks and war-graves in the area and general vicinity;
- wind farms;
- areas of heritage, nature conservation, archaeological and geological importance;
- recreational uses;
- general planning policies for the area (international, national, and local);
- any other legitimate use of the sea.

Evaluation of impacts

When evaluating the overall impact, it is necessary to identify and quantify the marine and coastal environmental consequences of the proposal. The EIA should evaluate the extent to which the proposed extraction operation is likely to affect other interests of acknowledged importance. Consideration should also be given to the assessment of the potential for cumulative impacts on the marine environment. In this context, cumulative impacts might occur as a result of aggregate dredging at a single site over time, from multiple sites in close proximity, or in combination with effects from other human activities (e.g., fishing and disposal of harbour dredgings).

It is recommended that a risk assessment be undertaken. This should include consideration of worst-case scenarios, and indicate uncertainties and assumptions used in their evaluation.

The environmental consequences should be summarized as an impact hypothesis. The assessment of some of the potential impacts requires predictive techniques, and it will be necessary to use appropriate mathematical models. Where such models are used, there should be sufficient explanation of the nature of the model, including its data requirements, its limitations, and any assumptions made in the calculations, to enable assessment of its suitability for the particular modelling exercise.

Mitigation measures

The impact hypothesis should include consideration of the steps that might be taken to mitigate the effects of extraction activities. These may include:

- the selection of aggregate dredging equipment and timing of aggregate dredging operations to limit impact upon the biota (such as birds, benthic communities, any particularly sensitive species and habitats, and fish resources);

- modification of the depth and design of aggregate dredging operations to limit changes to hydrodynamics and sediment transport and to minimize the effects on fishing;
- spatial and temporal zoning of the area to be authorized for extraction or scheduling extraction to protect sensitive fisheries or to respect access to traditional fisheries;
- preventing on-board screening or minimizing material passing through spillways when outside the dredging area to reduce the spread of the sediment plume;
- agreeing exclusion areas to provide refuges for important habitats or species, or other sensitive areas.

Evaluation of the potential impacts of the aggregate dredging proposal, taking into account any mitigating measures, should enable a decision to be taken on whether or not the application should proceed. In some cases it will be appropriate to monitor certain effects as the aggregate dredging proceeds. The EIA should form the basis for the monitoring plan.

Authorization issue

When an aggregate extraction operation is approved, then an authorization should be issued in advance (which may take the form of a permit, licence or other form of regulatory approval). In granting an authorization, the immediate impact of aggregate extraction occurring within the boundaries of the extraction site, such as alterations to the local physical and biological environment, is accepted by the regulatory authority. Notwithstanding these consequences, the conditions under which an authorization for aggregate extraction is issued should be such that environmental change beyond the boundaries of the extraction site are as far below the limits of allowable environmental change as practicable. The operation should be authorized subject to conditions which further ensure that environmental disturbance and detriment are minimized.

The authorization is an important tool for managing aggregate extraction and will contain the terms and conditions under which aggregate extraction may take place, as well as provide a framework for assessing and ensuring compliance.

Authorization conditions should be drafted in plain and unambiguous language and will be designed to ensure that:

a) the material is only extracted from within the selected extraction site;

b) any mitigation requirements are complied with; and

c) any monitoring requirements are fulfilled and the results reported to the regulatory authority.

Monitoring compliance with conditions attached to the authorization

An essential requirement for the effective control of marine aggregate extraction is the monitoring of dredging activities to ensure conformity with the authorization requirements. This has been achieved in several ways, e.g., an Electronic Monitoring System or Black Box. The information provided will allow the regulatory authority to monitor the activities of aggregate dredging vessels to ensure compliance with particular conditions in the authorization.

The information collected and stored will depend on the requirements of the individual authorities and the regulatory regime under which the permission is granted, e.g., EIA, Habitats, Birds Directives of the EU.

The minimum requirements for the monitoring system should include:

- an automatic record of the date, time, and position of all aggregate dredging activity;
- position to be recorded to within a minimum of 100 metres in latitude and longitude or other agreed coordinates using a satellite-based navigation system;
- that there should be an appropriate level of security;
- the frequency of recording of position should be appropriate to the status of the vessel, i.e., less frequent records when the vessel is in harbour or in transit to the aggregate dredging area, e.g., every 30 minutes, and more frequently when dredging, e.g., every 30 seconds.

The above are considered to be reasonable minimum requirements to enable the regulatory authority to monitor the operation of the authorization in accordance with any conditions attached. Individual countries may require additional information for compliance monitoring at their own discretion.

The records can also be used by the aggregate dredging company to improve utilization of the resources. The information is also an essential input into the design and development of appropriate environmental monitoring programmes and research into the physical and biological effects of aggregate dredging, including combined/cumulative impacts (see section above).

Environmental monitoring

Sand and gravel extraction inevitably disturbs the marine environment. The extent of the disturbance and its environmental significance will depend on a number of factors. In many cases, it will not be possible to predict, in full, the environmental effects at the outset, and a programme of monitoring may be needed to demonstrate the validity of the EIA's predictions, the effectiveness of any conditions imposed on the authorization, and therefore the absence of unacceptable impacts on the marine environment.

The level of monitoring should depend on the relative importance and sensitivity of the surrounding area. Monitoring requirements should be site-specific, and should be based, wherever possible, on the findings of the EIA. To be cost-effective, monitoring programmes should have clearly defined objectives derived from the impact hypothesis developed during the EIA process. The results should be reviewed at regular intervals against the stated objectives, and the monitoring exercise should then be continued, revised, or even terminated.

It is also important that the baseline and subsequent monitoring surveys take account of natural variability. This can be achieved by comparing the physical and biological status of the areas of interest with suitable reference sites located away from the influence of the aggregate dredging effects, and of other anthropogenic disturbance. Suitable locations should be identified as part of the EIA's impact hypothesis.

A monitoring programme may include assessment of a number of effects. When developing the programme, a number of questions should be addressed, including:

- What are the environmental concerns that the monitoring programme seeks to address?
- What measurements are necessary to identify the significance of a particular effect?
- What are the most appropriate locations at which to take samples or observations for assessment?
- How many measurements are required to produce a statistically sound programme?
- What is the appropriate frequency and duration of monitoring?

The regulatory authority is encouraged to take account of relevant research information in the design and modification of monitoring programmes.

The spatial extent of sampling should take account of the area designated for extraction and areas outside which may be affected. In some cases, it may be appropriate to monitor more distant locations where there is some question about a predicted nil effect. The frequency and duration of monitoring may depend upon the scale of the extraction activities and the anticipated period of consequential environmental changes, which may extend beyond the cessation of extraction activities.

Information gained from field monitoring (or related research studies) should be used to amend or revoke the authorization, or refine the basis on which the aggregate extraction operation is assessed and managed. As information on the effects of marine aggregate dredging becomes more available and a better understanding of impacts is gained, it may be possible to revise the monitoring necessary. It is therefore in the interest of all concerned that monitoring data are made widely available. Reports should detail the measurements made, results obtained, their interpretation, and how these data relate to the monitoring objectives.

Reporting framework

It is recommended that the national statistics on aggregate dredging activity continue to be collated annually by the ICES Working Group on the Effects of Extraction of Marine Sediments on the Marine Ecosystem (WGEXT).

Definitions

In these Guidelines, "marine sediment extraction" is intended to refer to the extraction of marine sands and gravels (or "aggregates") from the seabed for use in the construction industry (where they often directly replace materials extracted from land-based sources), and for use in flood and coastal defence, beach replenishment, fill and land reclamation projects. It is recognized that other materials are also extracted from the seabed, such as stone, shell materials, and maerl, and similar considerations to those set out in the Guidelines should also apply to them. The Guidelines do not apply to navigational dredging (e.g., maintenance or capital dredging operations).

In these Guidelines, the term "authorization" is used in preference to "permit" or "license" and is intended to replace both terms. The legal regime under which marine extraction operations are authorized and regulated differs from country to country, and the terms permit and license may have a specific connotation within national legal regimes, and also under rules of international law. The term "authorization" is thus used to mean any use of permits, licenses, or other forms of regulatory approval.

The ecosystem approach will be elaborated by further work in both OSPAR and ICES. The following definition has been used elsewhere: "the comprehensive integrated management of human activities based on best available scientific knowledge about the ecosystem and its dynamics, in order to identify and take action on influences which are critical to the health of marine ecosystems, thereby achieving sustainable use of ecosystem goods and services and maintenance of ecosystem integrity."

Revision of Guidelines

WGEXT will continue to review any new information, conclusions, and understandings from scientific research projects, and any reports from countries on their experiences with the implementation of the Guidelines and, where appropriate, will revise the Guidelines accordingly.

ANNEX 11: ACME/ACMP ADVICE BY TOPIC FOR THE YEARS 1992–2003

Numbers in the table refer to sections of the present report and of the ACMP or ACME reports from 1992 to 2003, in reverse chronological order. From 2001, relevant sections from the report of the Advisory Committee on Ecosystems (ACE) are also listed.
*Signifies major advice on that topic.

Topic	Sub-topic	2003	2002	2001	2000	1999	1998	1997	1996	1995	1994	1993	1992
Monitoring	Strategy									5.1	*4; *Ann. 1	5	5.1
	Programme evaluation				6.7						4.2		
	Statistical methods for design		4.3.3; Ann. 3	4.6.2; 4.6.3; 4.6.4	6.6.2	5.6							
	Benthos				6.8.1				6.1.2; 11.1; *Ann. 8				8; *Ann. 6
	NSTF/MMP											5.2	
	Sediments/guidelines		4.2.1	4.5.1; *Ann. 2			4.6; *Ann. 2	4.5; *Ann. 1	5.5; *Ann. 4		5.5	6.1; *Ann. 1	
	Sediment data normalization			4.5.1; *Ann. 2	6.5.1; Ann. 1	5.5		4.5.2	5.5.1		5.5		
	Sediment sensitivity, variance factors									5.6			
	Metals/sediments		4.2.1						9.5	5.6	5.5		
	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		
	Use of seabird eggs					13.2; Ann. 7	4.7.5						
	Spatial monitoring		4.3.3; Ann. 3	4.6.2; 4.6.4	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			4.4	6.3	5.2	4.4		5.1.3		5.4		
	Effects of nutrient enrichment		10.6			12.1			9.1	5.8			
	Monitoring PAHs						4.2; *Ann. 1	4.4.1; 4.5; *Ann. 1					
	Analytical and biological effects methods for priority substances	7.1	7.1										

Topic	Sub-topic	2003	2002	2001	2000	1999	1998	1997	1996	1995	1994	1993	1992
Assessment	Statistical methods	6.2; 6.3; Ann. 4,5	4.3; Ann. 1,2	4.6.1; Ann. 3,10			4.7.1						
	Combined effects of contaminants					5.1.5	11.2						
	Biological community data	10.1.2; Ann. 8		9.4									
	Data screening prior to assessment	5.6; 6.2.2	5.5; 5.6; Ann. 4,5	10.2; Ann. 8,9									
	Inputs and environmental concentrations	6.5	6.2										
Temporal trend monitoring	Strategy/objectives								*4; Ann. 1		*4; *Ann. 1		
	Guidelines								4.4		4; 5.2		
	Data analysis	6.2; 6.3; Ann. 4,5					4.7.4	15.2			5.2	6.2	5.2
	Nutrients								5.7			6.3	
	Fish/JMP/JAMP			4.6.5		5.6.4			5.6				
	Fish/CMP								5.6				
	Biota/BMP									7.3			
	Biological effects						4.1						
	Precision				6.6.5						5.2		*Ann. 1
	Sediments	4.3; Ann. 2	4.2.3						4.3; 5.5.3				
	Statistical requirements						4.7			4.3			
Integration of biological/ chemical measurements	Sediment quality	6.1; Ann. 3	4.2.2	6.3; 6.4					5.2.2	4.2; Ann. 2	5.4	6.4; *Ann. 2	
	Guidelines	4.2											
Biological effects monitoring	Monitoring strategy								*5.3	4.1; *Ann. 1			
	Statistical design						4.1*	4.31					
	Methods	4.1.2; Ann.1	4.1	4.1	6.1.2	5.1.1		4.3.2	5.3.2	Ann. 1			6.2
	Molecular techniques		4.1.2							*5.2			
	Pathology							4.3.3	5.3.3	8.4	9.4		
	Workshop results	4.1.1	4.1.1	4.1.1	6.1.4				5.3.2				6.1
	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	2003	2002	2001	2000	1999	1998	1997	1996	1995	1994	1993	1992
Baseline studies	Contaminants in ▪ Baltic sediments					5.2	4.3	6.1	7.1	7.1	7.1	8.1	13.2
	▪ North Sea sediments												13.1
	HCH in sea water												14
Regional assessments	Preparation plans/reviews			10.5	18.4; Ann. 9								
	North Sea QSR											4.1	4
	Baltic Sea								7.2	7.2	7.3		
	Baltic fish				18.2				7.3	7.2	7.3		
	AMAP		6.1										
Quality assurance	Reference materials			5.6; Ann. 4	7.5; Ann. 4		5.6	4.2			*6.9	7.11	
	Oxygen in sea water							*Ann. 3					
	Nutrients							5.7					
	Quality/comparability ▪ organic contaminants					5.4	4.5			*6.6	*6.8		
	Lipids									6.4	6.5		
	Biological effects techniques	5.3	5.3	5.3	7.3	7.4	5.4	5.3	*6.2; *Ann. 5	6.2		7.1	
	QA of sampling				7.7	7.6	5.7	5.10					*12.8
	QA info. In data bank			5.5				16.1.1			6.10		
	Chemical measurements–Baltic Sea	5.4	5.4	5.4	7.4	7.5	5.5	5.4	6.3	6.3	6.2	7.4	
	Biological measurements	5.1; 5.2	5.1; 5.2	5.1; 5.2	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			5.3			8.2	5.3.2	*Ann. 6				
	Use of QA data	5.6	5.5; 5.6 Ann. 4,5	10.2									
Intercomparison exercises	Status					Ann. 8		Ann. 10	Ann. 10	Ann. 7	Ann. 6	Ann. 5	Ann. 8
	Nutrients/sea water								6.4	6.5	6.6	7.8	12.4
	Hydrocarbons in biota									6.7			
	PAHs/standards												12.2
	PCBs/CBs in biota									6.6; 6.7	6.3;6.4	7.5	12.1
	Organochlorines in biota									6.6; 6.7			
	CBs in sediments									6.6	6.3	7.5	
	Metals in ▪ sea water							5.5	6.5				
	▪ SPM										6.7	7.9	12.3

Topic	Sub-topic	2003	2002	2001	2000	1999	1998	1997	1996	1995	1994	1993	1992
Intercomparison exercises (cont.)	Dissolved oxygen in sea water												12.5
	Oyster embryo bioassay												Ann. 4
	EROD												12.6; *Ann. 3
Methods	DO in sea water							*Ann. 3					
	Sediment normalization			4.5.1; *Ann. 2	6.5.1; Ann. 1						5.5		
	Organic carbon measurements							4.6; Ann. 2					
	Zooplankton studies		10.5; 10.6		6.8.3								
Phytoplankton and algal blooms	Primary production methods		10.6									6.5	11
	Dynamics of blooms	10.2					10.2	9.2			8	10	
	Exceptional blooms		10.4	9.3	6.8.2; Ann. 3	12.2; Ann. 2	Ann. 3	Ann. 8					
	Eutrophication effects on phytoplanktion		10.3										
Nutrients and eutrophication	Nutrients in the Baltic Sea		14.4										
	Nutrient trends/eutrophication in OSPAR area	14.3	14.3	11				Ann. 9					
	Nutrients and eutrophication						10.1	9.1	9.1	5.8		6.3	10
Fish diseases and related issues	Relation to pollution	8.3					8.2	8.3	5.3.3	8.4	9.4		
	Survey methods							7.2					
	Diseases in fish/shellfish	8.1	8.1	7.1	9.1	10.1							
	Baltic fish				9.4; *Ann. 6			7.2		8.1; 8.2; 8.3	9.3		
	Data analysis			7.2	9.2	10.2; Ann. 5	8.1; *Ann. 8	7.1	8.2		9.5	9.4	7
	M74 in Baltic salmon	8.2	8.2	7.1	9.3	10.3	8.3	6.2	7.4	7.4	9.1		
	Contaminants and shellfish pathology	8.3	8.5; Ann. 6										
	Diseases in farmed fish	8.1; 8.5	8.3; 8.4										
	Disease interactions between wild and farmed fish	8.4											

Topic	Sub-topic	2003	2002	2001	2000	1999	1998	1997	1996	1995	1994	1993	1992
Mariculture	Interactions		11.1; 11.3	12	14.1	15.2			15.1	14	13		9.1
	Guidelines for EIA		11.2.1										
	Monitoring		11.2.2										
	Escape of fish—effects	11.2			15.1		14.2	14.1					
	Nutrient inputs/Baltic				14.2								*9.2
	Use of chemicals		11.4		14.1.2	15.2		14.2					
	Implications of Water Framework Directive on mariculture	11.1; *Ann. 9											
Introductions and Transfers	Code of Practice	9.1; *Ann.7	9.2						14.2	13.1	14.1	12.1	
	Accidental transfers, including via ships	9.3; 9.4	9.3; 9.4	8.1.2; 8.3	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	9.2	9.1	8.1.1	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				
	Baltic marine mammal stocks	*ACE 2			*13.1	14.2		11.1	13.1		10.2		*18
	Populations/N. Atlantic	ACE 6.3		ACE 4.6							10.1	11.1	*18
	Pathogens											11.2; Ann. 3	
	Impact of fisheries	ACE 4; ACE 6.4	*ACE 2	ACE 3			12.1; *Ann. 9	11.2; 11.3	13.2				
	Monitoring programme for marine mammals in Baltic	*ACE 3											
Seabirds	Diet, food consumption					13.1; *Ann. 6							
	Use in contaminant monitoring			ACE 4.9		13.2; *Ann. 7				5.3			
	Effects of contaminants, plastic	ACE 6.5; 6.7; 6.8; 6.9			12.1								
	Populations	ACE 6.11		ACE 4.8									

Topic	Sub-topic	2003	2002	2001	2000	1999	1998	1997	1996	1995	1994	1993	1992
Overviews	Mercury						7.1; *Ann. 4						
	Hormone disruptors			6.2			7.4; Ann. 6		Ann. 2				
	Tris(4-chlorophenyl) methanol/methane		7.2.2	6.1.1	8.1				10; *Ann. 7				
	Benzene/ alkylated benzenes									10.2; *Ann. 5			
	Chlorinated alkanes				8.1					10.1; Ann. 4			
	PBBs and PBDEs	7.2.1	7.2.3	6.1.2		9.2; *Ann. 4							
	PCDDs and PCDFs		7.2.1	6.1.5; *Ann. 7									
	PCDEs							8.1; *Ann. 6					
	TBT					9.2; *Ann. 3							
	Phenylurea herbicides	7.2.2; *Ann. 6											
	Toxaphene		7.2.4					8.1; *Ann. 5			12.3		
	Atrazine										12.1		
	Irgarol 1051			6.1.3; *Ann. 5				8.1; *Ann. 4					
	Antifouling booster biocides			6.1.3 *Ann. 5									
	Volatile organic contaminants			6.1.4; Ann. 6									
Classification/ assessment tools	Hazardous substances	7.1	7.1				11.1			12.3			*15
	Background concentrations		6.4		6.2			15.1		12.1			
	Ecological Quality Objectives—North Sea	*ACE 6	6.5; *ACE 10	*ACE 4	18.6	17.2							
	Ecotoxicolgical reference values									12.2			
	Sediment quality criteria	6.1; *Ann. 3											
	Environmental indicators	6.4	6.6; 6.7	10.4		17.1							

Topic	Sub-topic	2003	2002	2001	2000	1999	1998	1997	1996	1995	1994	1993	1992
Sand/gravel extraction	Code of Practice/Guidelines	12.1; *Ann. 10	12.1; Ann. 7										
	Effects	12.2	12.2	13	16	8.1	6.1	*6.3					
	Environmental impact assessment	12.2	12.1; 12.3	13		8.1				*15	*15	13	
	Quantities extracted in Member Countries	12.2	12.2	13.1	16.1								
Modelling	Radioactive contaminants/Baltic Sea												*17.1
	Use in monitoring and assessment										16		17.2
Data banks and management	Nutrients	14.2	14.2	15.2	20.2	19.2	15.2	17.2	16.1.2				
	Contaminants	14.1	14.1	15.1	20.1	19.1	15.1	17.1	16.1.1; 16.3	17	2.2	2.2	
	NSTF												20
	ICES format		14.1; 14.6	15.1	20.1.10; 20.5	19.1			16.6				
	ICES databases	14.1.4	14	15.1	20.1	19.1						14	
	Biological database		14.5	15.3	20.3	19.3	15.1.3	17.3; 17.4			11.2; Ann. 4		
	AMAP			15.1	20.1.3	19.1.3		17.1.1	16.2				
Ecosystem effects of fishing	General	ACE 6.2			18.1			*12	12		18		*19
	Effects of disturbance on benthos, seabed habitats	ACE 9	*ACE 4		5		10.4	9.3	11.2	9; Ann. 3	11.1		8.3
	Seabird/fish interactions				12.2	4		10			19		
	Impact on non-target fish species	ACE 6.12	ACE 7				*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	ACE 6.12					13.1						
	Discards				*11		13.2						
	Effects on genetic diversity of exploited stocks	ACE 13	ACE 9										
	Ecosystem impacts of industrial fishing	*ACE 11											
	Effects on deep-sea cold-water corals	ACE 8	*ACE 3										

Topic	Sub-topic	2003	2002	2001	2000	1999	1998	1997	1996	1995	1994	1993	1992
Inputs of contaminants and nutrients	Riverine inputs (gross)						4.7.2; 4.7.3	*4.7.1					
	Trend detection methods	6.3; Ann. 5	4.3.1: Ann. 1,2	4.6.1; Ann. 3	*6.6.1; *Ann. 2	*6; *Ann. 1							
	Atmospheric inputs							4.7.1					
ICES Environmental Report	Oceanographic conditions	6.6.1	6.8.1	10.6.1	18.3.1	8.4.1	6.2.1						
	(Zoo)plankton	6.6.2	6.8.2	10.6.2			6.2.2						
	Harmful algal blooms	6.6.3	6.8.3	10.6.3	18.3.2	8.4.2; Ann. 2	6.2.3; Ann. 3						
	Fish disease prevalence	6.6.4	6.8.4	10.6.4	18.3.3; Ann. 8								
Special topics	Sediments—Baltic					5.2	4.3	6.1	7.1	7.1	7.1		
	Sediments (bioavailability)								9.3	4.2; Ann. 2	5.4	6.4; Ann. 2	
	Bioaccumulation of contaminants							8.2; *Ann. 7					
	Oil spill studies			Ann. 1									
	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/Project	10.1.1	10.1	9.1						9			*Ann. 5
	GLOBEC			14.1	19.1	18.1	16.2						
	GOOS	4.5	13.1	14.2	19.2	18.2	16.1	16					
	GIWA			14.3									
	Marine habitat classification/ mapping	ACE 7	ACE 5	ACE 5	17	16							
	Toxicity of dredged material	6.1; *Ann. 3			8.4								
	Potential ICES contribution to the implementation of the OSPAR JAMP	*13											
	OSPAR List: Threatened and Declining Species and Habitats	*ACE 5; Ann. 1	*ACE 6; Ann. 1										

Topic	Sub-topic	2003	2002	2001	2000	1999	1998	1997	1996	1995	1994	1993	1992
Special topics (cont.)	EcoQO for sensitive or opportunistic species	ACE 6.13											
	Relative importance of extrinsic factors on fish population dynamics compared to fishing	*ACE 10											
	Framework for the provision of integrated advice	ACE 15											

ACRONYMS

ACE	Advisory Committee on Ecosystems
AChE	acetylcholinesterase
ACME	Advisory Committee on the Marine Environment
ADMIS	Automated Mass Spectral Deconvolution System
ALA-D	δ-aminolevulinic acid dehydratase
AMAP	Arctic Monitoring and Assessment Programme
AMPS	Working Group on Analysis and Monitoring of Priority Substances and Pollution Control (EU)
AQC	analytical quality control
ASMO	Environmental Assessment and Monitoring Committee (OSPAR)
ASP	amnesic shellfish poisoning
BaPH	benzo[*a*]pyrene hydroxylase
BDE	bromodiphenylether
BDF	bioassay-directed fractionation
BDRF	Biological Data Reporting Format
BECPELAG	ICES/IOC Sea-going Workshop on Biological Effects of Contaminants in Pelagic Eosystems
BEEP	Biological Effects of Environmental Pollution in Marine Coastal Ecosystems
BEQUALM	Biological Effects Quality Assurance in Monitoring Programmes
BEWG	Benthos Ecology Working Group
BFRs	brominated flame retardants
CBs	chlorobiphenyls
CD-ROM	compact disc: read-only memory
CEA	cellular energy allocation
CEFAS	Centre for Environment, Fisheries and Aquaculture Science (UK)
CEMP	Coordinated Environmental Monitoring Programme (OSPAR)
COMBINE	Cooperative Monitoring in the Baltic Marine Environment (HELCOM)
CPR	Continuous Plankton Recorder
DGTs	diffusive gradients in thin films
DNA	deoxyribonucleic acid
DR-CALUX	dioxin-responsive chemical-activated luciferase gene assay
DSP	diarrhetic shellfish poisoning
DTA	Direct toxicity assessment
EC	European Commission
EcoQO	ecological quality objective
EEA	European Environment Agency
EIA	environmental impact assessment
ELISA	Enzyme-linked imunosorbent assay
EQS	environmental quality standard
ER-CALUX	oestrogen-responsive chemical-activated luciferase gene assay
ERL	effects range-low
ERM	effects range-median
EROD	ethoxyresorufin-*O*-deethylase
EQS	Environment Quality Standards
ETC/WTR	European Topic Center for Water
EU	European Union
GC	gas chromatography
GEF	Global Environment Facility
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
GIWA	Global International Waters Assessment
GLOBEC	Global Ocean Ecosystem Dynamics Programme
GMOs	Genetically Modified Organisms
GOOS	Global Ocean Observing System
GSI	gonado-somatic index
GST	Glutathione *S*-transferase
HAB	harmful algal bloom
HAEDAT	Harmful Algal Event Database
HBCD	hexabromocyclododecane
HELCOM	Helsinki Commission (Baltic Marine Environment Protection Commission)
HPLC	high-performance liquid chromatography
ICES	International Council for the Exploration of the Sea
IHNV	Infectious haematopoietic necrosis virus

IMO	International Maritime Organization
IOC	Intergovernmental Oceanographic Commission
IPNV	Infectious pancreatic necrosis virus
ISAV	Infectious salmon anaemia virus
ISO	International Organization for Standardization
ITIS	Integrated Taxonomic Information System
JAMP	OSPAR Joint Assessment and Monitoring Programme
LOD	limit of detection
LOESS	statistical smoother
LOICZ	Land-Ocean Interactions in the Coastal Zone
MCWG	Marine Chemistry Working Group
MON	Working Group on Monitoring (OSPAR)
MONAS	Monitoring and Assessment Group (HELCOM)
NAO	North Atlantic Oscillation
NORSEPP	ICES/EuroGOOS North Sea Ecosystem Pilot Project
NSBP	North Sea Benthos Project
NSBS	ICES North Sea Benthos Survey
OSPAR	OSPAR Commission
PAHs	polycyclic aromatic hydrocarbons
PBDDs	polybrominated dibenzo-*p*-dioxins
PBDEs	polybrominated diphenylethers
PBDFs	polybrominated dibenzofurans
PCBs	polychlorinated biphenyls
PEL	Probable Effects Levels
PGNSP	ICES-EuroGOOS Planning Group on the North Sea Pilot Project
PICES	North Pacific Marine Science Organization
POPs	persistent organic pollutants
PSP	paralytic shellfish poisoning
QA	quality assurance
QC	quality control
QPID	Quality Peak Identification and Database system
QSAR	quantitative structure-activity relationship
QUASIMEME	Quality Assurance of Information for Marine Environmental Monitoring in Europe
RIKZ	Rijksinstituut voor Kust en Zee [National Institute for Coastal and Marine Management]
SCCPs	short-chain chlorinated paraffins
SDV	Sleeping disease virus
SETAC	Society for Environmental Toxicology and Chemistry
SGBOSV	ICES/IOC/IMO Study Group on Ballast and Other Ship Vectors
SGBPI	Study Group on Modelling of Physical/Biological Interactions
SGGOOS	Steering Group on GOOS
SGSEA	Steering Group for a Sea-going Workshop on Pelagic Biological Effects Methods
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	Steering Group on Quality Assurance of Biological Measurements in the Northeast Atlantic
SPDV	Salmon pancreas disease virus
SPM	suspended particulate matter
SPMDs	semi-permeable membrane devices
SQC	Sediment Quality Criteria
SQGs	Sediment Quality Guidelines
SSML	sea surface microlayer
SSS	Society to Save the Sturgeon
TEL	Threshold Effects Levels
TIE	toxicity identification and evaluation
TIMES	*ICES Techniques in Marine Environmental Sciences*
UK	United Kingdom
U.S.	United States
USA	United States of America
VHSV	viral hæmorrhagic septicæmia virus
VTG	vitellogenin
WFD	Water Framework Directive
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
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
WGZE	Working Group on Zooplankton Ecology
YAS	Yeast Androgenic Screen
YES	Yeast Estrogenic Screen

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