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# Report of the ICES Advisory Committee on the Marine Environment, 1999

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31 May–5 June 1999


International Council for the Exploration of the Sea

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## ICES ADVISORY COMMITTEE ON THE MARINE ENVIRONMENT

**1998/1999**

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|---|---|---|
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| **Dr M. Héral** | | Chair, Mariculture Committee |
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| **Dr K. Brander** | | ICES GLOBEC Coordinator |
| **Dr A. Bjørge**[2] | | Chair, WGMMHA |
| **Dr S. Uhlig**[2] | | Chair, WGSAEM |

[1]Attended part-time. [2]Invited expert, attended part-time.

All Member Countries were represented at the meeting except Ireland (* not represented due to schedule conflict). Representation for Denmark was shared between Dr Pedersen and Mr Mellergaard.

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

| | |
|---|---|
| Dr J. Nørrevang Jensen | ICES Environmental Data Scientist |
| Ms M. Karlson | ICES Departmental Secretary |
| Ms M. Sørensen | ICES Environment Assistant |

## EXECUTIVE SUMMARY

The ICES Advisory Committee on the Marine Environment (ACME) met from 31 May to 5 June 1999 at ICES Headquarters in Copenhagen. As part of its work during this period, the ACME prepared responses to the requests made to ICES by the OSPAR Commission, the Helsinki Commission, and the European Commission DG XIV. This report contains these responses. In addition to responses to direct requests, this report summarizes the deliberations of ACME on topics for which advice was not directly requested but for which the ACME felt that there was information that would be of potential interest to the Commissions, ICES Member Countries, and other readers of this report.

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

#### Monitoring

In 1999, the ACME continued work on the development of biological effects monitoring programmes. The ACME has updated tables, originally presented in its 1997 report, containing lists of recommended and promising methods for monitoring biological effects of contaminants in the marine environment (Section 5.1.1). In addition, the results of a study of analytical and environmental sources of variability in biomarkers is included, based on a time series of EROD data (Section 5.1.2).

The ACME reviewed new information on statistical considerations relative to monitoring programmes (Section 5.6). Statistical methods for designing and assessing monitoring programmes are discussed in Section 5.6.1, covering in particular dynamic sampling strategies and the estimation of between-year variance components in temporal trend monitoring. An example to assist in determining the number of replicate samples to characterize an area in monitoring studies is provided (Section 5.6.2), based on the North Sea Benthos Study. In addition, examples of visualization tools for integrating and interpreting biological effects results are provided (Section 5.6.3).

In the context of identifying which contaminants can be monitored on a routine basis with adequate interlaboratory comparability, Section 5.4 lists nutrients that can be monitored in sea water, and organic contaminants and trace elements that can be monitored in biota, sediments, and sea water on a routine basis. This section also lists the lowest concentrations that can be monitored for each substance in each medium.

Finally, noting the widespread problem of maintaining long time series of monitoring due to cutbacks in funding, the ACME strongly emphasizes the importance of maintaining long-term monitoring of environmental conditions as a basis for integrated environmental assessments to support environmental and fisheries management systems (Section 5.7).

#### Quality Assurance and Intercomparison Exercises

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

For quality assurance work in relation to biological measurements in the Baltic Sea, the ACME noted that the five-year frequency of ring tests, recommended last year, appears to be too low to improve the quality of the data. Thus, at least annual ring tests should be organized (Section 7.1). Updating taxonomic checklists and maintaining good levels of taxonomic expertise are also essential for biological monitoring and require continuous attention.

For the OSPAR area, the ACME took note of draft guidelines for quality assurance (QA) of biological measurements under OSPAR, including a detailed summary of the general QA system for application to marine environmental sciences and critical QA factors for specific parameters to be monitored, i.e., chlorophyll *a*, phytoplankton, macrozoobenthos, and macrophytobenthos.

In terms of chemical measurements, further progress has been made in the development of additional technical annexes for the 'Guidelines on quality assurance of chemical measurements in the Baltic Sea', that were prepared in 1997 for the monitoring programmes carried out under the Helsinki Commission (Section 7.5). Technical notes on quality assurance procedures for the determination of dissolved oxygen, hydrogen sulphide, pH, and salinity in Baltic waters have now been completed.

**Overviews of Contaminants in the Marine Environment**

The ACME reviewed international and regional activities concerned with the identification of 'new' contaminants in the marine environment (Section 9.1). This is an important topic and will require new screening procedures for new contaminants, distinct from routine monitoring work.

The ACME considered a review note on the environmental distribution and effects of tributyltin, that was prepared with particular emphasis on aspects related to the continued use of TBT antifouling paints on large sea-going vessels (Annex 3). Based on the information contained in this review note, ICES recommends that the remaining uses of TBT on vessels larger than 25 m should be phased out as soon as practicable. ICES supports the recommendation made by the International Maritime Organization (IMO) Marine Environmental Protection Committee at its 42$^{nd}$ meeting that the application of organotin-based antifouling paints should be prohibited from 1 January 2003 and that they must be phased out altogether by 1 January 2008.

Information on polycyclic musk fragrances in the aquatic environment was considered, a summary of which is contained in Section 9.2. Further data need to be collected to determine the risks posed to marine organisms by these contaminants.

The ACME considered a paper on brominated flame retardants, which have become widely distributed in fish, seabirds, and marine mammals. There is an urgent need for information on the long-term toxicity of brominated flame retardants to marine organisms, and ICES recommends that research needs on this topic be urgently addressed by the marine science community. The paper on these substances is reproduced in Annex 4.

**Report sections responding to requests specific to the OSPAR Commission**

**Trend Assessment Tools for Data on Inputs of Contaminants**

A partial response to the very detailed OSPAR request for advice on trend assessment tools for input data is contained in Section 6 and Annex 1. This comprises responses to six requests concerning fine tuning of the statistical method used by OSPAR for inputs (termed 'trend-y-tector'), including advice on the smoother and the procedure for calculating the power of the trend tests (Section 6.2). Responses are also provided to six requests concerning the adjustment of input loads, to compensate for the effect of irregular fluctuations in inputs induced by varying flow rates, temperatures or precipitation (Section 6.3). Finally, a package of assessment tools is provided, with guidelines for the choice of appropriate methods and their use (Section 6.4). This package is general enough to be used when choosing methods for the assessment of temporal trends in a number of media, and is not limited to inputs of contaminants. Two implementation plans for the use of this material are provided. Work on this request will be finalized in 2000.

**Data Handling**

The annual review of data handling activities relevant to OSPAR requirements by the ICES Environmental Data Centre is contained in Section 19.1 of this report. Section 19.2 summarizes the work of the ICES Oceanographic Data Centre in handling nutrients data relevant to the OSPAR programmes. A brief review of the development of a biological community reporting format and database for phytoplankton, zooplankton, phytobenthos, and zoobenthos is given in Section 19.3.

**Report sections responding to requests specific to the Helsinki Commission**

**Baseline Study of Contaminants in Baltic Sea Sediments**

The ACME reviewed the history of ICES/HELCOM work on the study of sediments in the Baltic Sea, and particularly the outcome of an ICES/HELCOM workshop to discuss and present the results of the Baseline Study of Contaminants in Baltic Sea Sediments (Section 5.2). Some advice for the future continuation of sediment-related work in the Baltic Sea is included.

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

The ACME noted that, in order to handle this request, a number of relevant types of data must be collected and assessed. These data requirements are listed in Section 8.2. ICES Member Countries are requested to make their data on discards available for use in this work.

**Chapter on 'Baltic Fish Stocks, Diseases and Ecosystem Effects' for Fourth Periodic Assessment of the State of the Marine Environment of the Baltic Sea, 1994–1998**

A brief overview of the working groups contributing material to this chapter and the progress in this work is provided in Section 8.3. The final material will be compiled in spring 2000 to complete this chapter.

**Report section responding to requests specific to the European Commission DG XIV**

ICES prepared material in response to a DG XIV request to comment on whether measurable effects exist on sandeel predators, given the large quantities of sandeels that are being fished from the North Sea every year. This material reviews the main predators of sandeels in the northwestern North Sea and proposes that the breeding success of kittiwakes be used as the current best practical indicator of food availability at least to seabirds, and possibly to all sandeel-dependent predators in the vicinity of seabird breeding colonies. The detailed advice is contained in Section 4.

**Information on topics of general interest**

**Fish Diseases**

In reviewing information on diseases of marine fish and shellfish in the ICES area, the widespread occurrence of Viral Haemorrhagic Septicaemia (VHS)-like virus in various marine fish species in the Baltic Sea, the North Sea, and the eastern North Atlantic areas was noted. ICES Member Countries are encouraged to collate information on possible effects of this disease on wild fish populations. A marked increase in the prevalence of skin ulcers in Baltic cod was also noted; further studies on the aetiology of these ulcers should be conducted (Section 10.1).

As a follow up to the statistical analyses of disease prevalence data for dab and flounder conducted in 1998, the ACME reviewed the results of a case study of environmental, oceanographic, and fisheries data available in the ICES data banks that might be used to elucidate possible relationships between environmental factors and fish diseases. An overview of the results of this case study is contained in Section 10.2, with the details in Annex 5. On the basis of this work, ICES encourages Member Countries to enhance their efforts to submit historic and current data held in national data banks to the ICES data banks, to facilitate a more comprehensive holistic analysis of the interactions between natural and anthropogenic environmental factors and the health status of marine organisms.

Information was reviewed on the potential impact of pathogens in wild shrimp populations on cultured shrimp (Section 10.4.2). As the introduction of exotic viruses into culture facilities is a potentially great threat to the shrimp farming industry, particularly in the USA, the ACME requested that further information be collected on issues related to the transfer of these viruses.

**Effects of Extraction of Marine Sand and Gravel on Marine Ecosystems**

Available information on the quantities of marine sand and gravel extracted from the coastal regions in the ICES area is provided in Section 8.1.1. Reviews are also included of several national seabed resource mapping programmes and national approaches to environmental impact assessments in relation to extraction activities (Sections 8.1.2 and 8.1.3). A summary of the advantages and disadvantages of the main acoustic systems currently available for seabed mapping is provided in Section 8.1.4. Finally, some information is given on effects of marine aggregate extraction on marine biota (Sections 8.1.5 and 8.1.6).

**Introductions and Transfers of Marine Organisms**

The ACME reviewed information on accidental introductions and transfers of non-native marine species into the waters of ICES Member Countries (Section 11.1). A growing number of accidental transfers have now been documented, and some of the introduced species are causing significant problems in certain areas. In particular, the veined rapa whelk, which is capable of consuming commercial clams and oysters, has been introduced via ballast water to a number of areas, most recently Chesapeake Bay. There is potential for this species to have impacts on a global scale.

Issues relevant to the transfer of organisms via ships' ballast water and sediments are reviewed in Section 11.2. This material shows the need for continued international cooperation on ballast water management and control, so that the information and data needed to implement management strategies and better understand the mechanisms of transfer of organisms will be obtained on a comparable basis.

**Seabird Issues**

The ACME considered information on the diets of seabirds in the ICES area, as well as estimates of food consumption by seabirds in several parts of the ICES area. This information is summarized in Section 13.1, with further details on food consumption contained in Annex 6. In addition, the use of seabirds as monitors of marine contamination was reviewed in detail (Section 13.2.1 and Annex 7), with particular attention to the use of seabird eggs in monitoring organochlorines and mercury in the southern part of the North Sea (Section 13.2.2).

**Issues Related to Mariculture**

The ACME took note of progress in fishery and mariculture genetic research, particularly the development of genetically modified marine organisms (Section 15.1). In terms of environmental impacts of mariculture, information on the use of chemicals for sea lice control and the inadvertent transfer of non-native species in relation to the movement of live organisms for mariculture purposes was highlighted (Section 15.2).

**ICES Environmental Report**

Contributions to the ICES Environmental Report for 1999 have been made concerning oceanographic conditions (Section 8.4) and harmful algal blooms (Annex 3).

**Global Programmes**

The ACME reviewed recent activities by ICES for the North Atlantic in relation to the Global Ocean Ecosystem Dynamics (GLOBEC) programme (Section 18.1). Progress in the development of potential contributions of ICES to the Global Ocean Observing System (GOOS) were also considered, and it was noted that there are several ICES initiatives that are of direct relevance to GOOS (Section 18.2).

**Sources of Information Considered by the ACME at its 1999 Meeting**

At its 1999 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 |
| SGBWS* | ICES/IOC/IMO Study Group on Ballast Water and Sediments |
| SGMHM | Study Group on Marine Habitat Mapping |
| SGQAB* | ICES/HELCOM Steering Group on Quality Assurance of Biological Measurements in the Baltic Sea |
| SGQAC* | ICES/HELCOM Steering Group on Quality Assurance of Chemical Measurements in the Baltic Sea |
| SGQAE* | ICES/OSPAR Steering Group on Quality Assurance of Biological Measurements related to Eutrophication Effects |
| SGSEF | Study Group on Effects of Sandeel Fishing |
| WGAGFM | Working Group on the Application of Genetics in Fisheries and Mariculture |
| WGBEC | Working Group on Biological Effects of Contaminants |
| WGBFAS | Baltic Fisheries Assessment Working Group |
| WGEAMS | Working Group on Environmental Assessment and Monitoring Strategies |
| WGEIM | Working Group on Environmental Interactions of Mariculture |
| WGEXT | Working Group on the Effects of Extraction of Marine Sediments on the Marine Ecosystem |
| WGHABD | ICES/IOC Working Group on Harmful Algal Bloom Dynamics |
| WGITMO* | Working Group on Introductions and Transfers of Marine Organisms |
| WGMMHA | Working Group on Marine Mammal Habitats |
| WGMS | Working Group on Marine Sediments in Relation to Pollution |
| WGOH | Working Group on Oceanic Hydrography |
| WGPDMO | Working Group on Pathology and Diseases of Marine Organisms |
| WGPE | Working Group on Phytoplankton Ecology |
| WGSAEM | Working Group on Statistical Aspects of Environmental Monitoring |
| WGSE | Working Group on Seabird Ecology |
| WGSSO | Working Group on Shelf Seas Oceanography |
| WGZE | Working Group on Zooplankton Ecology |

Reports of the following other activities were also considered:

ICES/HELCOM Workshop on Baltic Sea Sediments: Conditions and Contaminants

ICES/HELCOM Workshop/Training Course on Phytoplankton (1998)

Workshop on GOOS

*These groups report directly to ACME.

# 1 INTRODUCTION

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

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

The ACME report is structured in terms of the topics covered at the ACME meeting on which it has prepared scientific information and advice; the topics include both those for which information has been requested by the Commissions or other bodies and those identified by the ACME to enhance the understanding of the marine environment. Information relevant to the Commissions' requests and specific issues highlighted by the ACME for their attention are summarized in Section 2 for the OSPAR Commission and Section 3 for the Helsinki Commission, where the individual work items from each Commission are listed and related to relevant sections of the main text. The full advice in response to the European Commission DG XIV request concerning potential impacts of sandeel fisheries on predator populations is contained in Section 4 of this report.

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

A summary of the progress on the 1999 programme of work requested by the OSPAR Commission is given below, along with reference to the relevant sections and annexes of this report where more detailed information can be found. This summary is provided according to the format of the Work Programme, with the questions on the Work Programme shown in italics and a summary of the ICES advice below in normal print.

## 1 SCIENTIFIC PEER REVIEW OF THE QSR 2000

*1.1 To arrange a special ACME meeting to conduct a scientific peer review of QSR 2000 with the following aims:*

*a. to establish the scientific veracity of the contents;*

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

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

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

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

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

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

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

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

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

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

At the request of OSPAR, the ACME meeting at which the scientific peer review of the OSPAR QSR 2000 was scheduled to take place has been postponed from November 1999 to January 2000. Thus, the response to this request will be published in the ACME report for the year 2000.

## 2 QUALITY ASSURANCE

*2.1 To continue to operate a joint ICES/OSPAR Steering Group on Quality Assurance of Biological Measurements related to eutrophication parameters (chlorophyll-a, phytoplankton, macrozoobenthos and macrophytobenthos) in order to coordinate:*

*a. the development of quality assurance procedures;*

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

*c. the preparation of appropriate taxonomic lists of species.*

*This work should cover the biological parameters within the eutrophication monitoring guidelines, namely: chlorophyll-a, phytoplankton, macrozoobenthos and macrophytobenthos. This is a fairly long-term programme (about five years) requiring the participation of scientists and technicians carrying out relevant analyses for this monitoring in laboratories in OSPAR Contracting Parties. Good cooperation should be ensured with the ICES/HELCOM Steering Group on Quality Assurance of Biological Measurements in the Baltic Sea.*

Information on progress in the work of the ICES/OSPAR Steering Group on Quality Assurance of Biological Measurements related to Eutrophication Effects (SGQAE) is contained in Section 7.2 of this report. SGQAE has prepared a draft document on guidelines for quality assurance of biological measurements under OSPAR/ICES, based on work carried out by the two ICES/HELCOM Steering Groups on quality assurance for the Baltic Sea, as well as other relevant material. This draft document is under review, with finalization intended for 2000. The ACME emphasizes that methodologies recommended in QA guidelines should be in accordance with methods currently accepted by ICES Working Groups, and also that recommendations should be identical for OSPAR and HELCOM, unless there is a clear reason for differences.

## 3 GUIDELINES

*3.1 To advise the Commission's appropriate subsidiary body about any changes required for the Commission's contaminant-specific or general biological effects monitoring guidelines, as a result of the work done on the development of the reporting formats.*

This request cannot be handled until the data reporting formats have been developed for all of the biological effects techniques adopted by OSPAR. The request for ICES to develop these reporting formats is contained in item 7, below, however OSPAR decided to fund this work over two years. Thus, the reporting formats for biological effects data will not be completed until some time in 2000, after which a review of any implications for the guidelines can be made.

## 4 ASSESSMENT TOOLS

*4.1 To continue the development of trend detection methods for input data in order:*

*a) to develop a method for including a power function in the trend detection methods for input data;*

*b) to develop provisions for the use of monthly data in the trend detection methods (taking into account INPUT(1) 98/4/1, §6.9, e.ii);*

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

This request has been specified in greater detail by the OSPAR Working Group on Inputs to the Marine Environment (INPUT), with advice requested under three groups of questions: 1) fine-tuning of the Trend-y-tector; 2) adjustment of loads; and 3) the use of monthly data (see full specification in Section 6.1). Detailed advice is contained in Section 6.2 concerning the fine-tuning of the Trend-y-tector and in Section 6.3 regarding the adjustment of loads. Technical details to support this advice are contained in Annex 1. In addition, Section 6.4 provides information on the identification of desirable components of a package of trend assessment tools and proposes a more general package of such tools, clearly identifying the role of individual components. Finally, two implementation scenarios are presented using this package of statistical tools; these implementations should satisfy the immediate requirements of OSPAR for the detection of trends in riverine and atmospheric inputs of contaminants.

Owing to the complexity of the task and time restrictions, the entire request could not be handled in one year. The remaining items will be completed in 2000.

## DATA HANDLING

*5 To carry out data handling activities relating to:*

*5.1 contaminant concentrations in biota and sediments;*

*5.2 measurements of biological effects;*

*5.3 the implementation of the Nutrient Monitoring Programme.*

The ICES Environmental Data Centre has handled all data submitted in 1998, covering monitoring activities in 1997. Further information on this work is contained in Section 19.1 of this report.

The ICES Oceanographic Data Centre continues to maintain as complete as possible a data set on nutrients and other oceanographic parameters in the ICES area. More details can be found in Section 19.2 of this report.

*6 To continue to establish databanks for:*

*6.1 phytobenthos, zoobenthos and phytoplankton species.*

Section 19.3 describes progress in the development of a Biological Data Reporting Format and data entry program for the submission of data on phytobenthos, zoobenthos, and phytoplankton species under the OSPAR Nutrient Monitoring Programme. The biological databases are being developed in association with this work. Review by potential users is scheduled for a six-month period beginning in late September 1999.

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

*Stage 1: the development of reporting formats for:*

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

*Stage 2: the development of reporting formats for:*

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

*Stage 3: the development of reporting formats for:*

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

A brief description of the work on this item, which is being funded over two years, is contained in Section 19.4 of this report. Most of the work on this request will be conducted in 2000, as information on quality assurance requirements for the above-mentioned methods is needed from the Biological Effects Quality Assurance in Monitoring Programmes (BEQUALM) project (see Section 7.4). It is anticipated that information from this project will become available before the end of 2000.

## 3 PROGRESS ON TASKS FOR THE HELSINKI COMMISSION

The present status of work on 1999 requests by the Baltic Marine Environment Protection Commission (Helsinki Commission) is given below, along with reference to the relevant sections and annexes of this report where more detailed information can be found. The requests are shown in italics and a summary of the ICES advice is then given in normal print.

**CONTINUING RESPONSIBILITIES**

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

The next evaluation of the status of populations of marine mammals in the Baltic Sea is scheduled for 2000. A brief summary of trends in populations of seals in the Baltic Sea is provided in Section 14.2 of this report.

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

Information on progress in the development of quality assurance procedures for biological measurements is summarized in Section 7.1 of this report. Further work has been completed on the development of quality assurance procedures for biological measurements; ICES proposes that this material be incorporated in the HELCOM COMBINE Manual. In addition, ICES recommends that regular ring tests be organized, preferably on an annual basis, for all laboratories that submit data for the COMBINE programme. There should be a system to coordinate ring tests, and to harmonize national and international ring tests. Furthermore, taxonomic training courses should be held on a regular basis for the expert staff from all laboratories submitting monitoring data to the HELCOM database.

With regard to quality assurance procedures for chemical measurements, progress is summarized in Section 7.5. The ACME accepted technical notes on 1) the determination of dissolved oxygen in sea water, 2) the determination of hydrogen sulphide, 3) pH measurement in sea water, and 4) the measurement of total alkalinity, for transmission to HELCOM for inclusion in the COMBINE Manual. In reviewing an evaluation report on the results of a questionnaire on laboratory performance of HELCOM laboratories, the ACME noted that for many of the parameters covered by the questionnaire, the evaluation of laboratory performance was hampered by the lack of clear analytical requirements in the COMBINE Manual. Accordingly, the ACME recommends that HELCOM develop more specific analytical requirements for all parameters in the COMBINE programme.

**SPECIAL ISSUES**

3) *To prepare a chapter on Baltic fish stocks, diseases and ecosystem effects for the Fourth Periodic Assessment of the State of the Marine Environment of the Baltic Sea, 1994–1998 comprising:*

   a) *an update of the material prepared by ICES for the Third Periodic Assessment concerning (i) commercial fish stocks, (ii) coastal fish, and (iii) diseases and parasites of Baltic fish; and*

   b) *information on the effects of mariculture (including the genetic effects of releases of cultured fish and possible use of genetically modified organisms); and*

   c) *information on the ecosystem effects of fishing activities in the Baltic Sea (taken from the material provided by ICES in 1997 in response to a HELCOM request, plus any new material);*

Material for this chapter is being compiled by the Baltic Fisheries Assessment Working Group (WGBFAS), the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO), the Working Group on Environmental Interactions of Mariculture (WGEIM), the Working Group on the Application of Genetics in Fisheries and Mariculture (WGAGFM), and the Working Group on Ecosystem Effects of Fishing Activities (WGECO). A brief review of the progress in the work is contained in Section 8.3 of this report.

4) *To arrange, together with HELCOM, a joint Workshop to discuss and present the results of the Baseline Study of Contaminants in Baltic Sediments;*

The ICES/HELCOM Workshop on Baltic Sea Sediments: Conditions and Contaminants was held in Helsinki on 14–16 April 1999. Information on the outcome of this Workshop, together with ICES advice on future monitoring of contaminants in sediments in the Baltic Sea, is contained in Section 5.2 of this report.

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

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

The ACME reviewed the data and assessment requirements to be able to answer this request and has planned activities so that this request can be handled in 2000.

*Request*

An EC DG XIV request, as follows: 'Large quantities of sandeels are being extracted from the North Sea every year. The question arises whether these quantities, although not harmful to the sandeel population themselves, might adversely affect predators of sandeels, like seabirds, marine mammals, and a number of fish species. ICES is requested to comment on whether measurable effects on sandeel predators exist. If this were the case, ICES is requested to comment on whether the establishment of closed seasons/areas for the sandeel fisheries could ameliorate these effects. ICES is requested to identify possible seasons/areas as specifically as possible.'

*Source of the information presented*

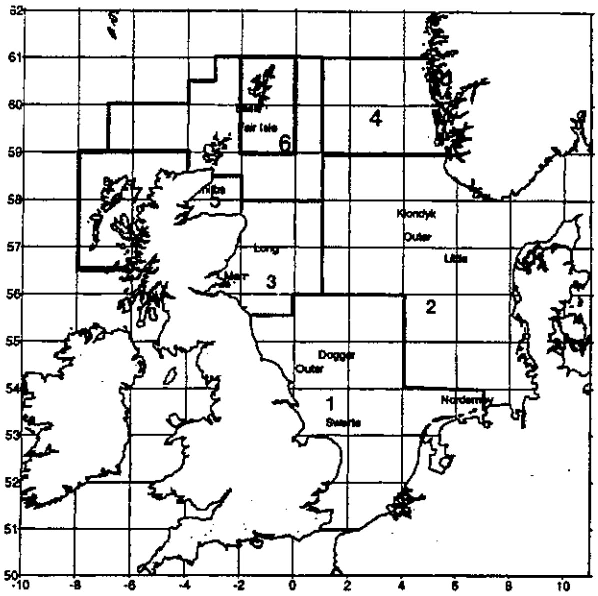
The report of the May 1999 meeting of the Study Group on Effects of Sandeel Fishing (SGSEF), the publications and Working Papers cited therein, and ACFM and ACME deliberations.

*Status/background information*

ICES first notes that although sandeels in the North Sea are assessed and managed as a single stock, recent research indicates that the North Sea 'stock' south of 58 °N can be broadly segregated into three regional stock units, as shown in Figure 4.1.

North of these three major areas and extending out of the North Sea to the west, four other regional stock units were identified: the Viking bank, Moray Firth, and the seas around the Northern Isles and Western Isles (Areas 4–7, respectively). Sandeels in the western part of Area 3 are of particular concern in the context of this request. They support a number of potentially sensitive seabird colonies, and the structure of this small sandeel stock unit appears to be particularly complex, with a number of small aggregations having limited exchange among them. Therefore, the individual sub-units may have a higher likelihood of recruitment overfishing than occurs in the larger sandeel population units. Assessment of the sandeel populations in the North Sea is only available for the entire North Sea stock complex. Estimates of sandeel abundance in individual areas, such as the area breakdown proposed for the North Sea (see Figure 4.1), may be useful in addressing more effectively the seabird-sandeel fisheries interactions, although even smaller areas may be required for this purpose.

**Figure 4.1.** Proposed sandeel stock structure (Areas 1–7), based on particle drift analysis (Procter *et al.*, 1998). The limits of the 'Shetland stock' are also shown.




1 = Dogger + southern banks 2 = Jutland, Fisher and Klondyke
3 = Wee bankie and Long 4 = Viking Bank 5 = Moray Firth
6 = Orkney and Shetland grounds 7 = Scottish west coast

In 1993 there was a measurable, negative effect of the fishery on the sandeel stock (local depletion) in the western part of Area 3, which coincided with a reduction in breeding success of seabirds, especially kittiwakes. Kittiwakes are thought to be a particularly sensitive indicator of the availability of sandeels to predators, because they have a high dependence on sandeels during the breeding season. (They nest on cliffs so their breeding success is little affected by the presence of terrestrial predators, and the foraging area overlaps that of the sandeel fishery.)

A single documented case of such a negative effect demonstrates that it is possible for sandeel fisheries to affect sandeel predators. In light of the long history of monitoring seabird breeding success in some colonies, a single observation suggests that such events are the exception rather than the rule. On the scale of the entire North Sea, the mortality of sandeels estimated to be caused by predators is more than three times the mortality caused by sandeel fisheries (see Figure 4.2), and the total consumption of sandeels by predators and industrial fisheries combined is estimated to have dropped greatly with the collapse of North Sea mackerel in the 1970s, and has varied relatively little thereafter (see Figure 4.3). However, catches in the industrial fisheries are concentrated in only certain parts of the North Sea (see Figure 4.4), so the possibility of local depletions of sandeels by fisheries exists. The effects of such depletions, were they to occur, would depend on whether or not the area hosted a population of predators dependent on sandeels.

**Figure 4.2.** Percentages of the total consumption of sandeels in the North Sea taken by North Sea mackerel, Western mackerel, gadoids, other fish and seals, industrial fisheries and seabirds, from 1974 to 1995. Data from ICES, 1997. From Furness (in press).

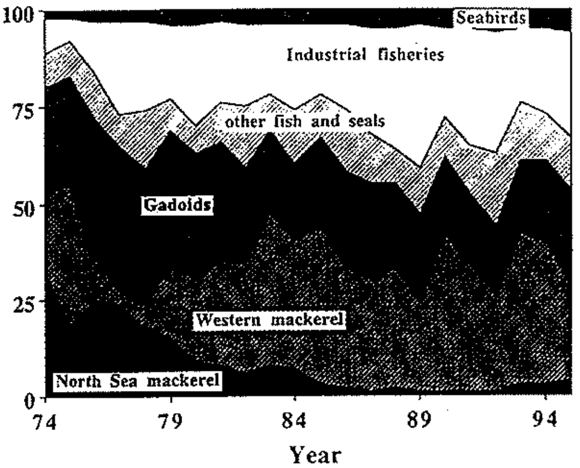



**Figure 4.3.** Estimated consumption of sandeels in the North Sea by major consumers (cod, mackerel, whiting, saithe, haddock, seabirds, seals, 'other fish predators', and industrial fisheries) showing trends over the study period from 1974 to 1995 (from ICES, 1997).

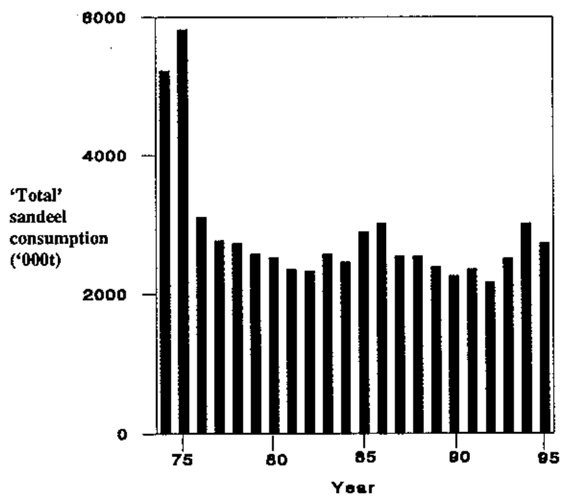
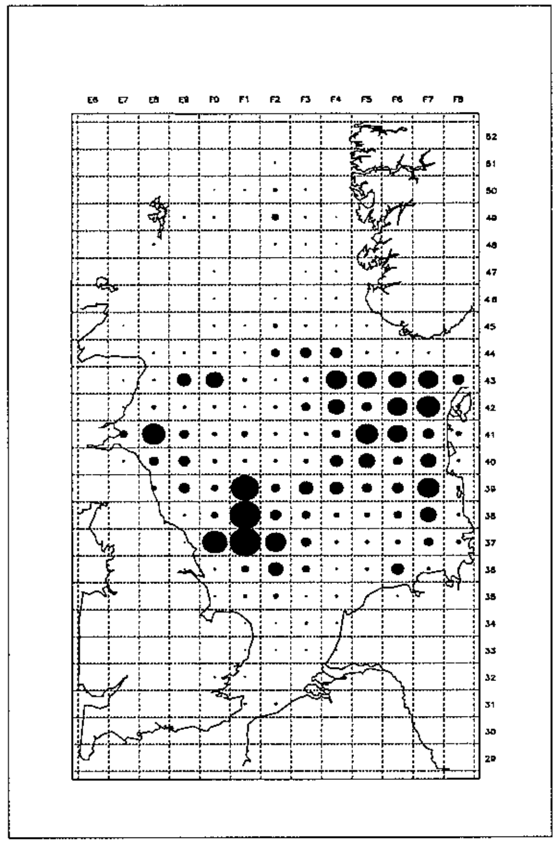



**Figure 4.4.** Relative geographical distribution of total international sandeel catches in 1994–1998 in the North Sea (data provided by Denmark and Norway).



The best data on distributions of predators on sandeels exist for seabirds during the breeding season. Moreover, the sensitivity of seabirds to prey abundance is particularly high during the breeding season, because the ability to change foraging sites in response to prey availability is constrained by the requirement to defend and care for nest sites and chicks.

Many seabird species that feed on small fish have a reproductive strategy in response to the large natural variability in their prey populations. The seabirds are long living and spread their reproductive effort over many years, producing a small number of eggs each year. Variable reproductive success of seabirds is therefore a natural phenomenon in response to variable abundance or availability of prey species.

A distinction must be made between abundance and availability of prey species for seabirds. The breeding success of seabirds depends on the match in time and space between the occurrence of prey of suitable size and behaviour and the feeding range of breeding seabirds. Low recruitment success of seabirds may reflect the mismatch in timing between prey availability and seabird feeding and not necessarily low abundance of the prey population.

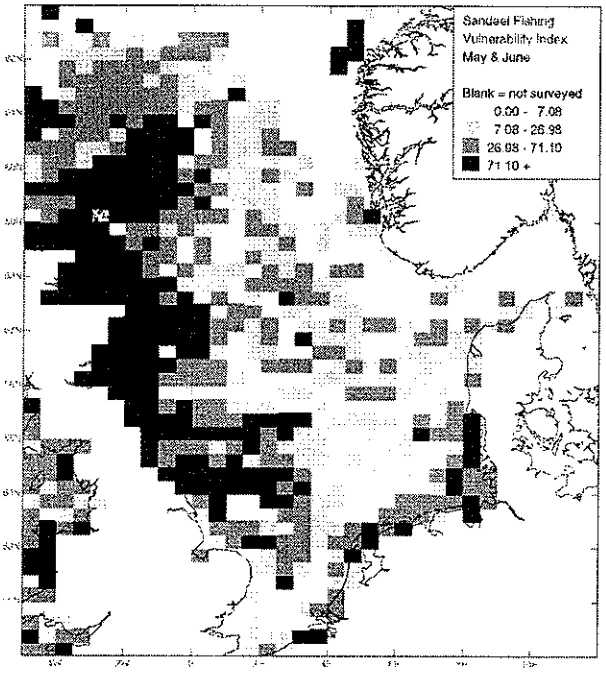
Despite the above comments, it is possible that kittiwake breeding success may be undesirably reduced in some locations owing to sandeel fisheries.

An index of sensitivity of seabirds to changes in sandeel abundance in May and June was therefore constructed, based on aspects of seabird foraging such as use of sandeels, energetic costs of foraging, diving ability, and ability to switch prey. This index shows that several coastal areas have high sensitivity values (see Figure 4.5). However, it must be stressed that the index is qualitative, and should be used for illustrative purposes but not prescriptive purposes. Moreover, the mapping procedure shows the 25 % of the North Sea with the highest sensitivity index scores, but 25 % was chosen arbitrarily, and it is not suggested that 25 %—or any other specific percent—of the North Sea warrants special protection.

The most important consideration is that empirically derived high sensitivity of seabirds to changes in sandeel abundance coincides with low breeding success of kittiwakes in the western part of Area 3.

Table 4.1 shows two examples comparing breeding success as given by the mean number of fledged kittiwake chicks per well-built nest (a standard measure used by ornithologists) over the last few years. The Shetland example suggests that low sandeel abundance may depress kittiwake breeding success. The Brittany data suggest that at < 0.5 fledged chicks per well-built nest kittiwake colonies decline. Furthermore, simulations using plausible values for population parameters of kittiwakes in the North Sea indicate that kittiwake populations will decline with a breeding success below 0.5 fledged chicks per well-built nest and increase with breeding success > 0.7 fledged chicks per well-built nest. Kittiwake breeding success in Area 3 to the west of 1 °W is currently (1996–1998, unpublished project data) at the low level of < 0.5 fledged chicks per well-built nest. In other areas, breeding success equals or exceeds 0.7 fledged chicks per well-built nest. In areas of high theoretical sensitivity other than Area 3 to the west of 1 °W, it is apparent that kittiwake breeding success is acceptable, but this situation should be prevented from deteriorating by discouraging the expansion of sandeel fishing effort in these areas.

**Figure 4.5.** Sensitivity of seabirds to changes in sandeel abundance in May and June. See text for explanation of index figures.




*Management Advice*

With regard to criteria for closure of areas to sandeel fisheries, with the objective of providing benefits to seabirds, ICES notes that several studies support the use of kittiwake breeding success as a surrogate measure for sandeel availability. There may, however, be important limitations on its generality as an indicator of sandeel availability to all predators. With this qualification, kittiwake breeding success is proposed as the best practical indicator of food availability at least to seabirds, and possibly to all sandeel-dependent predators in the vicinity of seabird breeding colonies. ICES **advises** using the criterion of kittiwake breeding success falling below 0.5 fledged chicks per well-built nest for three successive seasons as the threshold to close sandeel fisheries in areas important for foraging by the kittiwake colonies being monitored; the criterion of 0.7 fledged chicks per well-built nest for one year should be used to re-open the sandeel fishery thereafter.

**Table 4.1.** Breeding success of kittiwakes at Shetland and two colonies in Brittany used for comparative purposes.

| Chicks per nest (mean) | Location | Comment | Years | Reference |
|---|---|---|---|---|
| 1.04 | Brittany, France[1)] | Flourishing kittiwake colony | 1980–1985 | Danchin and Monnat, 1992 |
| 0.62 | Shetland[2)] | Moderate sandeel stock | 1991–1997 | Furness, 1999 |
| 0.49 | Brittany, France[1)] | Declining kittiwake colony | 1980–1985 | Danchin and Monnat, 1992 |
| 0.32 | Shetland | Low sandeel stocks | 1986–1990 | Furness, 1999 |

[1)]Different colonies.
[2)]Sandeel fishing prohibited in Shetland during 1991–1994.

Applying these criteria, ICES **advises** that the sandeel fishery west of 1 °W in sandeel Area 3 (see Figure 4.1) be closed to sandeel fishing (except as specified below), because the recent (over the past three years) breeding success of kittiwakes has been less than 0.5 fledged chicks per well-built nest. Such a closure should stay in force until kittiwake breeding success exceeds 0.7 fledged chicks per well-built nest. ICES also **advises** that during the period of closure a very limited commercial monitoring fishery be conducted, not to exceed 10 boat-days in each of May and June. The objective of this limited fishery is to maintain a time series of CPUE and biological sampling data on sandeels in this area, so participating vessels should be representative of the fleet operating in that area in recent years. ICES therefore also **advises** that the fishery in the remainder of sandeel Area 3, east of 1 °W, should not be allowed to exceed the recent average harvest in that portion of Area 3 which would remain open, approximately 30 000 t.

The sandeel fishery in sandeel Areas 1 and 2, as it has operated in the past, has not caused any documented problems for either the sandeel stock units or their predator populations. However, the closure and catch restrictions advised above for part of sandeel Area 3 are likely to displace effort. ICES therefore further **advises** that management measures be implemented to ensure that concentrations of fishing effort do not build up in excess of past practice of the fishery in sensitive areas identified in sandeel Areas 1 and 2.

Data are insufficient in the other sandeel areas (Areas 4–7) to evaluate effects of sandeel fisheries on sandeels and their predators. However, ICES **advises** that measures be taken to ensure that concentrations of effort not be allowed to build up in these areas. ICES also **advises** that any expansion of current fisheries in these areas should be accompanied by monitoring of kittiwake productivity, or other suitable surrogates for food availability to dependent predators.

**References**

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# 5 MONITORING GUIDELINES AND TECHNIQUES

## 5.1 Biological Effects Monitoring

### 5.1.1 Recommended and promising biological effects monitoring techniques

*Request*

There is no specific request; this updates information presented in the 1997 ACME report and is part of continuing ICES work to review and evaluate marine biological effects monitoring techniques.

*Source of the information presented*

The 1999 report of the Working Group on Biological Effects of Contaminants (WGBEC) and ACME deliberations.

*Status/background information*

The ACME last dealt with this subject two years ago, and published lists of recommended and promising marine biological effects monitoring techniques in its 1997 report (ICES, 1997).

The ACME noted that in 1999 the WGBEC revisited the issue and added some new techniques to the 1997 lists. In particular, the following recommended methods have now been included:

1) oxidative stress indicators in fish (these are not contaminant specific, but respond to a wide range of contaminants by measuring the presence of damaging free radicals);
2) early toxicopathic lesions have been added to the range of histopathological endpoints that can be used in fish liver tissue (the whole package of liver histopathology measures genotoxic and non-genotoxic changes associated with exposure to a range of contaminants, including PAHs, nitro-organics and triazines);
3) protein or enzyme altered foci in fish (these also respond to PAHs, nitro-organics and triazines, and indicate that exposure to carcinogens has occurred).

The list of promising methods requiring more research now includes the following additions:

a) the COMET assay (so-called because it can reveal a 'comet's tail' of DNA fragments from a stressed cell) for measuring genotoxic effects in cell cultures exposed to sediment extracts;
b) measurement of apoptosis (programmed cell death) in fish liver;
c) the use of enzyme-linked immunosorbent assays (ELISA) for measuring DNA adducts caused by genotoxins;
d) DR-CALUX (dioxin-responsive chemical-activated luciferase) and ER-CALUX (oestrogen-responsive chemical-activated luciferase) reporter gene assays for compounds active via the aryl hydrocarbon (Ah)-receptor and oestrogen-receptor, respectively.

Approximately sixty new references to published information on biological effects methods were added to the lists. The revised list of recommended methods is presented in Table 5.1.1.1 and the revised list of promising methods is contained in Table 5.1.1.2.

**Table 5.1.1.1.** Recommended techniques for biological monitoring programmes at the national or international level.

| Method | Organism | Refs. | Currently used in monitoring programmes | Quality control | Issues addressed | Biological significance |
|---|---|---|---|---|---|---|
| Bulky DNA adduct formation | Fish[1]<br>Bivalve molluscs | 1–6, 157–159 | F, NL, S, USA | B[2] | PAHs<br>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. |
| Acetylcholinesterase (AChE) inhibition* | Fish[1]<br>Crustacea<br>Bivalve molluscs | 12–16, 114, 116, 118 | F | | Organophosphates and carbamates or similar molecules<br>Possibly algal toxins | Measures exposure. |
| Metallothionein induction | Fish[1] | 17–22, 173 | MEDPOL, N | B[2] | Measures induction of metallothionein protein by certain metals (e.g., Zn, Cu, Cd, Hg) | Measures exposure and disturbance of copper and zinc metabolism. |

*Intercomparisons or quality control procedures complete for some methods (e.g., Refs. 31, 40, 99, 100).
[1]May also be applicable to mammals and birds. [2]Quality control under BEQUALM.
F=France; MEDPOL=Monitoring and Research Programme of the Mediterranean Action Plan; N=Norway; NL=Netherlands; S=Sweden; USA=United States.

**Table 5.1.1.1.** Continued.

| Method | Organism | Refs. | Currently used in monitoring programmes[‡] | Quality control | Issues addressed | Biological significance |
|---|---|---|---|---|---|---|
| Ethoxyresorufin-*O*-deethylase (EROD) or cytochrome P4501A induction* | Fish[1] | 46–51, 99, 115 | D, F, NL, UK, B, MEDPOL, N | B[2] | Measures induction of enzymes which detoxify planar organic contaminants (e.g., PAHs, planar PCBs, dioxins) | Possible predictor of pathology through mechanistic links. Sensitive indicator of present exposure. |
| δ-aminolevulinic acid (ALA-D) inhibition | Fish[1] | 74–75 | N | B[2] | Lead | Index of exposure. |
| Oxidative stress indicators | Fish[1] | 76–78 | | | Not contaminant specific, will respond to a wide range of environmental contaminants | Measures the presence of free radicals. |
| Fluorescent bile metabolites | Fish | 79–80 | N, NL, UK | | PAHs | Measures exposure to and metabolism of PAHs. |
| Lysosomal stability | Fish[1]<br>*Mytilus* spp. | 23–25 | D, MEDPOL | B[2] | Not contaminant specific, but will respond 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. |
| Early toxicopathic lesions, pre-neoplastic and neoplastic liver histopathology | Fish[1] | 7–11, 108, 110, 119–130, 164–167 | D, NL, UK, USA | | PAHs<br>Other synthetic organics, e.g., nitro-organics, amino triazine pesticides (triazines) | Measures pathological changes associated with exposure to genotoxic and non-genotoxic carcinogens. |
| Whole sediment bioassays* | • *Corophium*<br>• *Echinocardium*<br>• *Arenicola*<br>• *Leptocheirus*<br>• *Grandidierella*<br>• *Rhepoxynius*<br>• *Ampelisca* | 31–35 | NL, UK, USA, CAN | B[2] | Not contaminant specific, will respond to a wide range of environmental contaminants in sediments | Acute/lethal and acute/sublethal toxicity only at present. May enable retrospective interpretation of community changes. |
| Sediment pore water bioassays* | Any water column organism including:<br>• *Dinophilus*<br>• sea urchin fertilization, etc.<br>• bivalve embryo<br>• Microtox | 36–41 | F, NL, USA | | Will respond to a wide range of environmental contaminants | Acute and chronic toxicity, including genotoxicity, etc. Toxicity of hydrophobic contaminants might be underestimated in pore water assays. |
| Sediment sea water elutriates* | Any water column organism, as above | 36–41 | NL, UK | | Will respond to a wide range of environmental contaminants in<br>• dredge spoils<br>• sediments liable to resuspension | Acute/lethal and acute/sublethal toxicity, including genotoxicity, etc. |
| Water bioassays* | As for pore water and elutriates (see above) | 36–41 | NL, UK, USA, CAN | B[2] | Not contaminant specific, will respond to a wide range of environmental contaminants in inshore and estuarine waters | Acute/lethal and acute/sublethal toxicity, including genotoxicity, etc. |
| Scope for growth* | Bivalve molluscs, e.g., *Mytilus* | 55–58 | | | Responds to a wide variety of contaminants | Integrative response which is a sensitive and sublethal measure of energy available for growth. |

*Intercomparisons or quality control procedures complete for some methods (e.g., Refs. 31, 40, 99, 100). [1]May also be applicable to mammals and birds. [2]Quality control under BEQUALM. [‡]B=Belgium; CAN=Canada; D=Germany; F=France; MEDPOL=Monitoring and Research Programme of the Mediterranean Action Plan; N=Norway; NL=Netherlands; S=Sweden; UK=United Kingdom; USA=United States.

**Table 5.1.1.1.** Continued.

| Method | Organism | Refs. | Currenty used in monitoring programmes‡ | Quality control | Issues addressed | Biological significance |
|---|---|---|---|---|---|---|
| Shell thickening | *Crassostrea gigas* | 103 | Portugal | | Specific to organotins | Disruption to pattern of shell growth. |
| Vitellogenin induction | Male and juvenile fish | 26–30 | N, UK | | Oestrogenic substances | Measures feminization of male fish and reproductive impairment. |
| Imposex | Neogastropod molluscs, e.g., dogwhelk (*Nucella lapillus*) | 52–54, 174 | CAN, D, Ireland, Iceland, N, NL, S, UK | B[2] Q[3] | Specific to organotins | Reproductive interference. Estuarine and coastal littoral waters (*Nucella*) and offshore waters (*Buccinum*). |
| Intersex | Littorinids | 101, 102 | D, Ireland, N, UK | B[2] | Specific to reproductive effects of organotins | Reproductive interference in coastal (littoral) waters. |
| Protein or enzyme altered foci | Fish | 92, 144–150 | | | PAHs<br>Other synthetic organics, e.g., nitro-organics, amino triazine pesticides (triazines) | Indicates exposure to carcinogen(s). |
| Reproductive success in fish | • *Zoarces viviparus*<br>• *Pseudopleuronectes americanus*<br>• *Gadus morhua* | 72, 153, 160 | D, S, USA, N | B[2] | 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. In viviparous fish, restricted to period when young are carried by female. |
| Externally visible fish disease | Fish | 104–108, 168–171 | CAN, D, DK, UK, USA, NL, B | | Measures the effects of non-specific stress by quantifying the presence of externally visible diseases, especially in dab (*Limanda limanda*) | These diseases are natural, but may be exacerbated by various stressors, including contaminants. |
| Benthic community analysis* | Macro-, meio-, and epibenthos | 42–45, 100, 109 | B, CAN, D, F, Ireland, N, UK, USA, NL | B[2] | 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. |

*Intercomparisons or quality control procedures complete for some methods (e.g., Refs. 31, 40, 99, 100). [2]Quality control under BEQUALM. [3]Quality control under QUASIMEME. ‡B=Belgium; CAN=Canada; D=Germany; DK=Denmark; F=France; N=Norway; NL=Netherlands; S=Sweden; UK=United Kingdom; USA=United States.

**Table 5.1.1.2.** Promising biological effects monitoring methods that require further research before they can be recommended for monitoring.

| Method | Organism | Refs. | Issues addressed | Biological significance |
|---|---|---|---|---|
| DNA strand breaks | Fish and mussels | 113 | Not contaminant specific, will respond to a wide range of environmental contaminants | Measures genotoxic effects, but is also extremely sensitive to other environmental parameters. |
| Oncogenes | Fish | 93–95 | PAHs<br>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. |
| Cytochrome P4501A induction | Invertebrates | 96 | Induced enzyme response to PAHs, planar PCBs, dioxins and/or furans | Measures exposure to organic contaminants. |
| Glutathion-*S*-transferase(s) (GST) | Fish, mussels | 97, 154 | Predominantly organic xenobiotics | Measures exposure and the capacity of the major group of Phase II enzymes. |
| Multidrug/multixenobiotic resistance (MDR/MXR) | Fish, invertebrates | 85–92, 131–143 | Organic xenobiotics | Measure of exposure. |

**Table 5.1.1.2.** Continued.

| Method | Organism | Refs. | Issues addressed | Biological significance |
|---|---|---|---|---|
| Various methods of measuring immunocompetence | Fish, invertebrates | 73 | Not contaminant specific, will respond to a wide range of environmental contaminants | Measures factors which influence susceptibility to disease. |
| On-line monitoring | Mussels, crabs | 98 | Responds to metals and xenobiotics | Measures the effects of chemicals on heart rate using a simple and inexpensive remote biosensor. Gives an integrated response. |
| Degenerative gill and kidney histopathology | Fish (especially flatfish such as dab (*Limanda limanda*)) | 59–66 | General toxicological response which will respond to a wide variety of contaminants | Measures degenerative change in tissues. |
| Abnormalities in wild fish embryos and larvae | Many fish, including demersal and pelagic species | 70–71, 172 | Not yet linked unequivocally to contaminants | Measures frequency of probably lethal abnormalities in fish larvae. Mutagenic, teratogenic. |
| Chronic whole sediment bioassays | Invertebrates | 32 | 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. |
| Pollution-induced community tolerance (PICT) water bioassay | Microalgae | 67–69 | Specific contaminants can be tested | Measure of degree of adaptation to specific pollutants. Not yet widely tested. |
| COMET assay (*in vitro* bioassay for sediments) | Cells exposed to extracts | 111, 155, 156 | Genotoxic compounds | Genotoxic potential of sediments. |
| Apoptosis | Fish | 112 | Responds to a wide range of contaminants | Research state. |
| Enzyme-linked immuno-sorbent assay (ELISA) for DNA adducts | Fish | 161–163 | Not contaminant specific | Genotoxic effects. |
| Dioxin-responsive chemical-activated luciferase gene assay (DR-CALUX) | Cells exposed to samples or extracts | 151 | Aryl hydrocarbon (Ah) receptor active compounds | Possible predictor of pathology. |
| Oestrogen-responsive chemical-activated luciferase gene assay (ER-CALUX) | Cells exposed to samples or extracts | 152 | Oestrogen receptor active compounds | Potential endocrine disruption. |
| Allometric response in the benthic community | Macro-, meio-, and epibenthos | 81–84 | Not contaminant specific, will respond to a wide range of environmental contaminants | Ecosystem level. Retrospective. |

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*Need for further research*

Further research is required to investigate the methods considered as showing promise that are listed in Table 5.1.1.2.

*Recommendation*

ICES endorses the use of the recommended biological effects monitoring techniques listed in Table 5.1.1.1.

**Reference**

ICES. 1997. Report of the Advisory Committee on the Marine Environment, 1997. ICES Cooperative Research Report, 222: 12–20.

### 5.1.2 Analytical and environmental sources of variability in biomarkers in fish and implications for monitoring guidelines

*Request*

There is no specific request; this is part of the continuing ICES work on the development of sampling designs for biological effects monitoring.

*Source of the information presented*

The 1999 reports of the Working Group on Biological Effects of Contaminants (WGBEC), the Joint Meeting (JBSAEM) of WGBEC and the Working Group on Statistical Aspects of Environmental Monitoring (WGSAEM), and ACME deliberations.

*Status/background information*

In connection with the development of sampling designs for biological effects monitoring, the ACME emphasized in 1997 the importance of (a) having clear objectives (e.g., the detection of spatial distributions or temporal trends), (b) specifying the relevant variables for measurement (e.g., cytochrome P4501A induction), (c) identifying important covariables (e.g., sex, season), and (d) identifying the sample size and number of samples required to attain the requisite statistical power for testing the hypothesis in question. In addition, the ACME has advocated for several years the integration of biological and chemical monitoring (ICES, 1997).

The existing advice was further elaborated in 1998 with the provision of information on the design of effective sampling schemes for a single biological effects variable, a suite of biological effects variables, and methods for integrating a suite of biological effects variables into a meaningful index of impact. In order to stimulate the further development of statistical sampling designs, the ACME endorsed the work programme, proposed by the 1998 joint meeting of WGBEC and WGSAEM, to develop integrated sampling designs for the entire range of biomarkers and biological effects endpoints that are relevant to the OSPAR Joint Assessment and Monitoring Programme (JAMP) and other marine monitoring programmes in the ICES area (ICES, 1999).

In addition, WGSAEM reviewed the requirements on biomarker data with regard to their statistical analysis and commented on the following aspects:

1) If the interest is to reduce variability and to allow an association between values in the fish and its surroundings, observations should be made on a weakly migrating species.

2) As each species requires its own analysis, restriction to one species increases the power of the analysis, given a fixed budget.

3) If the interest is to model the relationships within individuals, the parameters of interest should be measured preferably within one individual.

4) If the interest is to explain why individuals behave differently, host-specific data should be reported (sex, length, age, condition indicators, spawning status, etc.).

5) If the comparison of sites is intended, all data should be collected at the same time. Site-specific parameters that might explain why relationships might differ from site to site should be reported (temperature, salinity, contaminants outside the fish, etc.).

6) If the temporal comparison of relations within one site is intended, data should be collected as above; host-specific (sex, age, etc.) and external parameters (temperature, salinity, etc.) that might be responsible for temporal changes should be reported.

7) If time lags play a role, because, e.g., a biomarker is expected to react on an exposure only after a certain time span, then data should be recorded at several points in time so that the expected time lag is covered.

The identification of analytical and environmental sources of variability in biomarkers in fish and their implications for monitoring guidelines addresses the objective of developing integrated sampling designs by aiming to collate and interpret the many pieces of information necessary for designing an integrated monitoring programme, in particular, to explore the power of a temporal trend monitoring programme in accordance with the four ACME objectives. In order to achieve this, a long time-series data set was provided by the CLO-Department of Fisheries, Ostend, Belgium, which consisted of measurements of hepatic EROD activity and related biotic and abiotic variables in dab (*Limanda limanda*) from the southern North Sea. Sampling began in 1992 with at least four sampling periods per year, except for 1998 and 1999 when only data for March were available. Most samples were collected from the Belgian continental shelf, but on several occasions other locations in the southern North Sea were visited. Research aspects have included investigations of:

- the seasonal relationships between EROD levels, liver fat, and the concentrations of PCBs and PAHs;
- the relationships between EROD levels and covariates such as dab size, sex, and sexual development.

The information is summarized in two sections below. Section 5.1.2.1 presents a summary of the data set and the main findings of the study. Section 5.1.2.2 uses subsets of the data to estimate within-year and between-year variance components, and to explore the power of a temporal trend monitoring programme on EROD levels. The estimates were used to assess the effectiveness of the current JAMP sampling guidelines for temporal trend monitoring. Due to the nature of the exercise, the seasonal relationships between EROD levels, liver fat, and the concentrations of PCBs and PAHs were not considered.

### 5.1.2.1 Summary of study on EROD levels in dab

The main findings of the study are summarized below.

**Seasonal variation**

EROD activity, liver fat, and PCB concentrations were measured in the same liver of dab in samples taken from the Belgian continental shelf in 1992 and 1993. PAH concentrations were measured in other dab due to the insufficient size of the livers sampled. The following results thus cover a period of two years.

The dab showed seasonal changes in their endogenous biochemical and exogenous chemical compositions. Liver fat concentrations were low in winter and 2.5- to 3-fold higher during the second half of the year. The seasonal fat profile paralleled that of the seasonal water temperature profile. Liver fat was clearly used as an energy source in the coldest part of the year (February/March). Fat consumption is probably initiated by the combined effects of a decreased food supply and decreased feeding activity in winter. Fat production and storage in the liver (an important fat-storing organ in dab) started with rising water temperatures.

The seasonal profile of PCB concentrations, measured on a wet weight basis, paralleled the fat profile. Low PCB concentrations were present in early spring, and the highest concentrations (2.5- to 3-fold higher) were measured in autumn at the same time as the greatest liver fat concentrations were found. On a fat weight basis, the PCB concentrations were approximately constant over the sampling period. Important removal and uptake fluxes of PCBs took place within a year.

The seasonal variation in EROD levels was inversely related to the seasonal variation in liver fat and liver PCB concentrations. High EROD levels were found when liver fat and PCB concentrations were low. EROD levels dropped to often non-detectable levels during the uptake of PCBs by the liver. The seasonal cycles suggest that the high EROD levels in early spring may be (in part) due to the removal of PCBs from the liver during fat metabolism. Thus, these peaks may be triggered by natural modulators (e.g., temperature), but caused by fluxes in PCBs. It was stressed that the PCBs measured in this study are not the most potent inducers of EROD. However, it is known that planar PCBs follow similar seasonal patterns because all PCB congener concentrations remain approximately proportional in dab (de Boer *et al.*, 1993).

PAHs could not be detected in the livers at the low to sub-ppb level. It is therefore assumed that PAHs made little contribution to the induction of EROD. Salinity levels were approximately constant throughout the year, so salinity is also unlikely to have influenced EROD levels.

In connection with these results, a computer simulation was noted showing how the basal EROD activities of fish exhibit different seasonal patterns in different parts of the North Sea (Lange *et al.*, 1995). For North Sea dab, EROD activities are relatively low in autumn, and show strongly increased values around the spawning season, with sex-dependent differences during the time when peak activities are found. However, the spawning season of dab varies within the North Sea, with a shift in time from the English Channel to the northern North Sea. The spawning season also varies to some extent from year to year, depending on climatic conditions. There is a general consensus that temperature conditions are a key factor influencing the gonadal cycle and the time of maturation. As a result, the seasonal patterns of basal EROD activities in dab vary according to the different temperature regimes in different parts of the North Sea. Clearly, spatial variability in EROD activity can only be interpreted with a good understanding of the eco-physiological parameters triggering these seasonal variabilities.

**Relationship between EROD levels, body size, and sexual development**

These results concern the relationship between EROD levels, body size, and sexual development in dab sampled from the Belgian continental shelf in March. They thus relate to the period of peak EROD activity.

In both females and males, EROD levels were approximately constant in juvenile and small dab (body size ≤ 10 cm), but decreased rapidly with increasing body size thereafter.

Sex hormones are known to be potent inhibitors of cytochrome P450 activities (Hansson and Gustaffson, 1981; Stegeman *et al.*, 1982; Vodicnik and Lech, 1983; Jiminez and Stegeman, 1990). In this study, female EROD levels decreased sharply with increasing ovary size, defined as the length between the basis and top of the ovary triangle. For females of the same body size, there was on average about a 50 % difference in EROD levels between those with no noticeable gonads and those with 12 mm ovaries. Ovary size is easy to measure, and including information about this covariate might help in the design of an EROD monitoring programme. A corresponding, easy-to-measure indicator of the reproductive state of males was not found.

For both sexes, the relationship between EROD levels and gonadosomatic index (GSI) or gonad weight was less clear. The determination of both GSI and gonad weight cannot be done onboard ship and requires the storage of samples before weighing on land. Disadvantages of the use of GSI and gonad weight are:

- part or all of the liver must be removed for biochemical measurement (and thus will not be available for weighing on land);
- the effects of storage (which causes loss of weight by drying);
- the selection of dab to avoid the effects of maturation on EROD cannot be done before the EROD determinations have been made (as they must be made immediately onboard ship).

**Relationship between EROD levels and PAH concentrations**

These results concern the relationship between EROD levels and PAH concentrations at different sites at times of the year outside the 'early spring peak'.

On the Belgian continental shelf, low to non-detectable EROD levels were generally observed, and PAHs could not be detected in the livers at the low to sub-ppb level. However, higher EROD levels were observed in areas where dab contained detectable PAH concentrations in the liver. For example, significantly higher EROD levels were found close to an oil drilling platform at the Rhine field in Dutch coastal waters. Here, PAH concentrations were detectable, but did not exceed 18 ng Σ PAHs/g liver (wet weight), which suggests that EROD is sensitive enough to signal the presence of very low levels of PAHs. In this case, elevated EROD levels in the second half of the year are likely to be specifically caused by the effects of PAHs.

### 5.1.2.2 Variance components for EROD

EROD data subsets used to estimate the within-year and between-year variance components were extracted for two periods, March and autumn, and excluded as much as possible interferences by confounding factors.

**Within-year and between-year variance components for log-EROD in March**

The data subset for March was chosen because:

- it provided the most data for analysis: fish collected from the same place at approximately the same time of year over six years;
- seasonal effects could be ignored;
- it minimized the difficulties with zero measurements (below detection limits).

Within-year and between-year variance components for log-EROD were estimated for dab in the length range 20–25 cm. Males and females were treated separately. This was in keeping with the JAMP guidelines for temporal trend monitoring of EROD, which call for analysis of between 10 and 20 dab of the same sex (from the same place at the same time of year) within the size range 20–25 cm (OSPAR, 1998). For comparison, the same variance components were also estimated for dab in the size range 15–20 cm.

Before the evaluation was completed, data collected in March 1999 were made available, so the variance estimates were updated. Variance components were also estimated for the size ranges < 10 cm and 10–15 cm. For each sex and size class, Table 5.1.2.2.1 gives:

1) the number of years with at least one observation $T$;
2) the minimum and maximum values of $n_t$, the number of observations within each year (excluding those years where there were no data at all);
3) estimates of the within-year and between-year variances, $\sigma_w^2$ and $\sigma_y^2$, of log-EROD, with approximate standard errors (s.e.);
4) estimates of the within-year and between-year relative standard deviations, $100\sigma_w$ and $100\sigma_y$; these give (approximately) the % coefficient of variation of EROD measurements on the original scale (i.e., before log-transformation).

**Table 5.1.2.2.1.** Variance components for EROD in dab collected in March, according to sex and size class.

| Sex | Size (cm) | $T$ | $n_t$ min | $n_t$ max | $\sigma_w^2$ est | $\sigma_w^2$ s.e. | $100\sigma_w$ | $\sigma_y^2$ est | $\sigma_y^2$ s.e. | $100\sigma_y$ |
|---|---|---|---|---|---|---|---|---|---|---|
| Male | < 10 | 4 | 11 | 41 | 0.36 | 0.04 | 60 | 0.10 | 0.09 | 32 |
| | 10–15 | 4 | 25 | 73 | 0.59 | 0.06 | 77 | 0.07 | 0.07 | 26 |
| | 15–20 | 5 | 1 | 154 | 0.56 | 0.05 | 75 | 0.69 | 0.54 | 83 |
| | 20–25 | 4 | 2 | 31 | 0.66 | 0.15 | 81 | 0.98 | 0.94 | 99 |
| Female | < 10 | 4 | 15 | 77 | 0.52 | 0.08 | 72 | 0.00 | 0.01 | 0 |
| | 10–15 | 5 | 3 | 50 | 0.69 | 0.09 | 83 | 0.00 | 0.01 | 0 |
| | 15–20 | 7 | 4 | 74 | 1.43 | 0.15 | 120 | 0.50 | 0.36 | 71 |
| | 20–25 | 5 | 5 | 41 | 1.57 | 0.24 | 125 | 0.95 | 0.77 | 97 |

**Table 5.1.2.2.2.** The power of detecting a 10 % or 20 % change in EROD levels per year over 10 years at the 5 % significance level for 10 or 20 dab sampled each year, according to sex and size class.

| Sex | Size (cm) | 10 % yearly change 10 dab | 10 % yearly change 20 dab | 20 % yearly change 10 dab | 20 % yearly change 20 dab |
|---|---|---|---|---|---|
| Male | < 10 | 0.58 | 0.64 | 0.99 | 0.99 |
| | 10–15 | 0.60 | 0.99 | 0.99 | 0.99 |
| | 15–20 | 0.15 | 0.16 | 0.46 | 0.47 |
| | 20–25 | 0.12 | 0.13 | 0.35 | 0.36 |
| Female | < 10 | 0.93 | 0.99 | 0.99 | 0.99 |
| | 10–15 | 0.86 | 0.99 | 0.99 | 0.99 |
| | 15–20 | 0.17 | 0.19 | 0.51 | 0.56 |
| | 20–25 | 0.12 | 0.12 | 0.33 | 0.35 |

The within-year variances are estimated reasonably well. EROD levels in 'large' females (> 15 cm) appear to be more variable than EROD levels in 'small' females (< 15 cm) or in males. This could be due to sex- and size-specific differences in the effects of maturation during the spawning season.

The between-year variances are estimated rather poorly. The large standard errors here do not indicate that the between-year variances are zero, merely that the estimates are only of the correct order of magnitude. This reflects the difficulty of estimating between-year variances with relatively short time series. In particular, the suggestion that the between-year variance is small (negligible) for smaller fish **should be treated with caution and not be over-interpreted.** There was some consideration of the number of years required to obtain 'adequate' estimates of between-year variance. However, WGSAEM has shown that between-year variances are not well estimated even with ten years of data. The practical solution is to combine estimates from several (many) shorter time series.

The variance components in Table 5.1.2.2.1 were used to estimate the power of a temporal monitoring programme on EROD levels in dab, following the JAMP sampling guidelines. The theory is given in Nicholson *et al.* (1997). Table 5.1.2.2.2 shows the power of detecting a 10 % or 20 % change in EROD levels per year over ten years at the 5 % significance level, having sampled either 10 or 20 individual dab each year.

Again, **these results should not be over-interpreted,** due to the lack of precision in the between-year variance estimates. Note that increasing the sample size from 10 to 20 individuals each year has a negligible impact on power. The poor power is attributable to the large between-year variances.

Clearly, this work needs to be reinforced with comparable estimates from other studies. However, it is also arguable that monitoring EROD during the spawning season is fraught with problems of interpretation and should be avoided, as specified in the JAMP guidelines.

**Within-year and between-year variance components in autumn**

Because of the difficulties in interpreting EROD levels in dab collected during the spawning season, some time was spent investigating variance components for EROD levels in dab collected in autumn. To obtain a reasonable number of years of data, EROD levels from September, October, and November were pooled. These levels are shown by sex, year, and length class in Figure 5.1.2.2.1, plotted on a fourth root scale to reduce skewness. Unfortunately, there are many zero measurements (below limits of detection), so the techniques used for the March data set, above, cannot be employed.

Modelling EROD data with a non-negligible quantity of zero values presents problems. One possible approach would be to model the data using some mixture distribution. For example, a binomial distribution could be used to model the number of zero measurements, and a gamma or log-normal distribution used to model the positive measurements. However, this poses several methodological problems, and was not pursued further. A simpler approach is to consider the proportion of EROD measurements above some threshold value, rather than model the EROD measurements themselves. With an appropriate threshold, this can be regarded as measuring the proportion of individuals with 'significantly induced' EROD activity. Provided that the threshold value is above the detection limit, an individual EROD measurement can always be classified as above or below the threshold. Given this framework, it is possible to estimate appropriate variance components, and to consider, e.g., the power to detect an increase over time in the proportion of individuals above the threshold.

To illustrate, consider the data for dab in the size class 15–20 cm (since these give the most complete time series), and let the threshold be 50 pmol resorufin min$^{-1}$ mg$^{-1}$ protein.

First, some notation: Let $n_t$ be the number of male (female) dab in the size class 15–20 cm sampled in year $t$, and let $y_t$ be the number having EROD levels above the threshold. If $p_t$ is the probability that a male (female) dab in year $t$ has EROD levels above the threshold, then $y_t$ can be assumed to have a binomial distribution

$$y_t \sim \text{Binomial}(n_t, p_t).$$

Table 5.1.2.2.3 shows the values of $n_t$, $y_t$, and estimates of $p_t$, by sex and year.

There is no evidence of any systematic trend in the probabilities $p_t$, but they clearly vary over time, particularly for females. A reasonable assumption, analogous to that made for log-EROD levels in March, is that there is random between-year variation in the probabilities $p_t$.

**Figure 5.1.2.2.1.** Plot of EROD levels in autumn by sex (males at the top of the plot, females on the bottom), year, and length class. Note that the plotting symbols are jittered to show the many zero measurements. EROD is plotted on a fourth root scale to reduce skewness. The vertical line in each panel corresponds to the threshold of 50 pmol resorufin min$^{-1}$ mg$^{-1}$ protein.

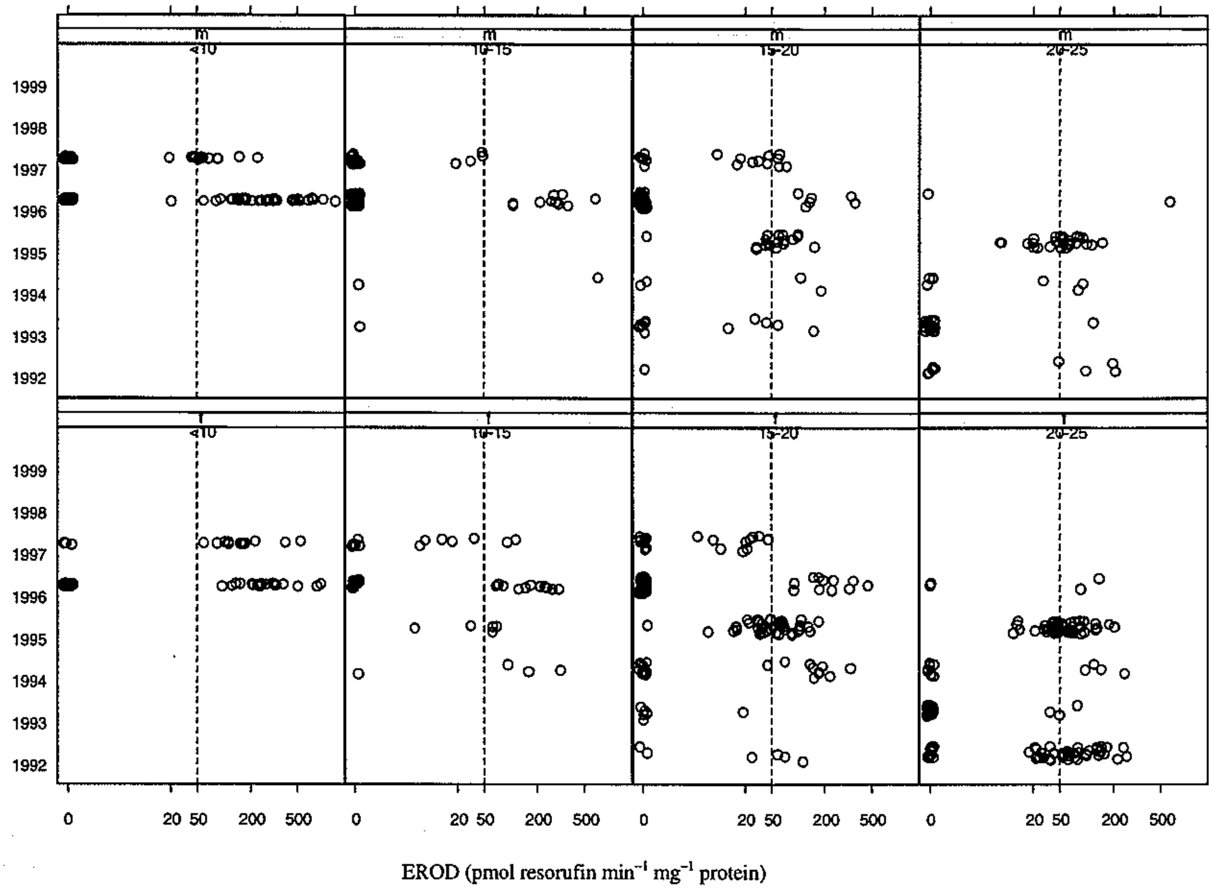

**Table 5.1.2.2.3.** Values of $n_t$, $y_t$, and estimates of $p_t$ by sex and year for dab in size class 15–20 cm sampled in year $t$.

| Year | Females | | | Males | | |
|---|---|---|---|---|---|---|
| | $n_t$ | $y_t$ | $p_t$ | $n_t$ | $y_t$ | $p_t$ |
| 1992 | 6 | 3 | 0.50 | 1 | 0 | 0.00 |
| 1993 | 6 | 0 | 0.00 | 12 | 2 | 0.17 |
| 1994 | 19 | 9 | 0.47 | 4 | 2 | 0.50 |
| 1995 | 38 | 21 | 0.55 | 19 | 11 | 0.58 |
| 1996 | 39 | 11 | 0.28 | 51 | 6 | 0.12 |
| 1997 | 19 | 0 | 0.00 | 17 | 4 | 0.24 |

Generalized linear mixed models provide a sensible statistical framework for modelling the data. These models cope naturally with binomial data, and allow estimation of a between-year variance component (and also of any systematic trends). The within-year variance is pre-determined by the properties of the binomial distribution to be unity (although this can often be relaxed, if required).

Assuming no systematic trend, the following generalized linear mixed model was fitted to the data for each sex. As before, the observations $y_t$ are assumed to have a binomial distribution

$$y_t \sim \text{Binomial}(n_t, p_t).$$

In addition, the probabilities $p_t$ are assumed to vary stochastically as

$$\text{logit}(p_t) = \log\left(\frac{p_t}{1-p_t}\right) \sim \text{N}(\mu, \sigma_y^2).$$

Here,

- the logit (or logistic) transformation is used to avoid the problems of probabilities being less than zero or greater than unity;
- $\mu$ is the underlying probability, on the logistic scale, that an EROD measurement is above the threshold;
- $\sigma_y^2$ is the between-year variation in the probability that an EROD measurement is above the threshold, again on the logistic scale.

The table below gives the estimates of $\mu$ and $\sigma_y^2$ by sex, with approximate standard errors (s.e.).

| | $\mu$ | | $\sigma_y^2$ | |
|---|---|---|---|---|
| | **estimate** | **s.e.** | **estimate** | **s.e.** |
| Females | –1.08 | 0.66 | 2.03 | 1.62 |
| Males | –1.01 | 0.47 | 0.76 | 0.80 |

The means $\mu$ are on the logistic scale. Back-transforming them shows that, on average, about 25 % of female EROD measurements and about 27 % of male EROD measurements are above the threshold.

Now consider the power of a monitoring programme to detect a temporal *increase* in the proportion of individuals above the threshold. Suppose that in each year $t = 1 \ldots T$, EROD activity is measured in $n$ male (female) dab in the size class 15–20 cm. Define $y_t$ and $p_t$ as before, but now suppose that

$$\text{logit}(p_t) = \log\left(\frac{p_t}{1-p_t}\right) \sim \text{N}(\mu_t, \sigma_y^2),$$

with

$$\mu_t = \mu_1 + \beta(t-1).$$

Here,

- $\mu_t$ is the underlying probability, on the logistic scale, of being above the threshold in year $t$,
- $\mu_t$ increases linearly over time;
- $\beta$ measures the strength of the trend.

Suppose the focus is on detecting an increase in EROD activity. Appropriate null and alternative hypotheses might then be:

$H_0$: $\beta \le 0$

$H_1$: $\beta > 0$.

These hypotheses can be tested by estimating $\beta$ using a generalized linear mixed model and then comparing $\hat{\beta}/\text{s.e.}(\hat{\beta})$ to the upper $100(1-\alpha)$ % quantile of a Normal distribution. Here,

- $\hat{\beta}$ is the estimate of $\beta$,
- $\text{s.e.}(\hat{\beta})$ is the standard error of $\hat{\beta}$,
- $\alpha$ is the size of the test.

Table 5.1.2.2.4 gives the power of a ten-year programme with 10 (20) individuals sampled each year to detect an increase in EROD activity at the 5 % significance level. The power depends on both $\beta$ and $\mu_1$. Here, $\mu_1$ is taken to be −1.0 (i.e., a back-transformed proportion of 27 %) reflecting the current status quo, and the power has been tabulated for different values of $\beta$. To aid interpretation, the back-transformed value of $\mu_{10}$ (i.e., the proportion of individuals above the threshold in the last year of monitoring) is also shown. The power must be calculated by simulation, and each value in the table is based on 100 independent realizations. With this number of realizations, the table is only a rough guide to the true power of the programme, but the salient features about the joint effects of $n$, $\beta$, and $\sigma_y^2$, are still clear.

Thus, if $\beta = 0.3$, for example, the proportion of individuals above the threshold will have increased from 27 % to 85 % over the ten-year monitoring period. Sampling 10 females each year gives an approximate power of 45 %. Increasing to 20 females each year increases the power to approximately 54 %. For males, the corresponding powers are 77 % and 79 %. The increased power for males is because the male between-year variance is lower. Note that increasing the sample size from 10 to 20 individuals per year only gives a small increase in power for the levels of between-year variance observed here.

*Need for further research or additional data*

Noting that a minimum of ten years of data is required to obtain adequate power, the ACME emphasized that a debate is needed concerning the magnitude of change that should be able to be detected and the time frame over which such a change should be detected.

The ACME noted that the method presented above appears to be promising, but investigation of other EROD data sets and other threshold values would be worthwhile.

If temporal monitoring is the main focus of interest, then there is a clear need to extend the above work by collating variance estimates from other studies, both for EROD in dab, and for other fish species and biomarkers. For example, analysis of the data in the ICES database indicated that only a few data were gathered at the same station in a series of years and that the month of sampling and the measurement technique varied over time, so these data cannot be used to estimate temporal variance components.

In the short term, there is a need to investigate the variation, and correlations, in a suite of biological, chemical, biomarker, and endpoint measurements in fish sampled from the same place and at the same time, that is, the within-sample variation. Several data sets exist that could be used for this purpose.

In the medium term, temporal variance components should be able to be estimated using data collected under the new JAMP guidelines. These data should both form consistent time series and provide information about correlations between several biological, chemical, biomarker, and endpoint measurements made on the same individuals.

In addition to the above discussion, the ACME noted that WGBEC has approved a proposal to conduct a pilot survey of cytochrome P450 (EROD) activity in juvenile dab. Monitoring programmes such as the OSPAR JAMP currently require EROD to be measured in adult dab, but there is evidence that the cycle of sexual maturity is a major confounding variable which can interfere with the interpretation of biological effects caused by contaminants (see Section 5.1.2.1, above). These problems might be avoided by using dab in the length range 5–9 cm, which also have the advantage of being less mobile than adults, so EROD induction would be more likely to reflect local exposure. However, before the routine use of juveniles can be recommended as an addition to monitoring EROD in adults, a more complete understanding is needed of the relationships between cytochrome P450 activity and dab age, size and maturity state in different parts of the ICES area. One problem with using small fish is that the desirable aim of integrating a number of measurements of different endpoints from single individuals cannot be met, as there may not be sufficient tissue available for the different assays. Pilot surveys of EROD induction in juvenile dab are therefore required to investigate the feasibility of the more routine use of juvenile dab in monitoring programmes.

**Table 5.1.2.2.4.** The power of a 10-year programme, with 10 and 20 individuals sampled each year, to detect an increase in EROD activity at the 5 % significance level.

| $\beta$ | $\mathrm{logit}^{-1}(\mu_{10})$ | Females | | Males | |
|---|---|---|---|---|---|
| | % | $n = 10$ | $n = 20$ | $n = 10$ | $n = 20$ |
| 0.0 | 27 | 0.11 | 0.08 | 0.01 | 0.03 |
| 0.1 | 47 | 0.18 | 0.12 | 0.25 | 0.26 |
| 0.2 | 69 | 0.25 | 0.39 | 0.53 | 0.51 |
| 0.3 | 85 | 0.45 | 0.54 | 0.77 | 0.79 |
| 0.4 | 93 | 0.70 | 0.68 | 0.88 | 0.97 |
| 0.5 | 97 | 0.83 | 0.90 | 0.96 | 0.99 |

*Recommendations*

ICES strongly encourages Member Countries to submit data on biological effects measurements for inclusion in the ICES database, particularly where these data form a consistent time series, in order to allow the further development of sampling schemes for biological effects monitoring.

ICES also encourages Member Countries to conduct pilot surveys on the relationships between cytochrome P450 activities and the size, age, and maturational state of juvenile dab (*Limanda limanda*) in the length range 5–9 cm in different parts of the ICES area. Juvenile dab possess the highest hepatic EROD level-to-noise ratio and are considered more sedentary compared to older specimens. For these reasons, juvenile dab may be useful for spatial and temporal trend monitoring purposes. The proposed pilot studies are needed because EROD measurements in adult dab are subject to serious confounding factors that limit their value. It is therefore necessary to investigate the relationships of cytochrome P450 activities with the size, age, and maturational state of juvenile dab throughout the ICES area to establish whether these dab are a more appropriate subject for routine monitoring.

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### 5.1.3 Possible expert systems for the evaluation of biological effects monitoring data

*Request*

There is no specific request; this is part of continuing ICES work to improve the evaluation and interpretation of marine monitoring data.

*Source of the information presented*

The 1999 reports of the Working Group on Biological Effects of Contaminants (WGBEC) and the Joint Meeting (JBSAEM) of WGBEC and the Working Group on Statistical Aspects of Environmental Monitoring (WGSAEM), and ACME deliberations

*Status/background information*

The ACME noted that in 1998 WGBEC had identified the possibility of using so-called expert systems to evaluate complex marine monitoring data sets (consisting of both biological and chemical determinands). These data sets are becoming increasingly multifactorial and, correspondingly, more difficult to assess. For example, data are now being collected under the OSPAR Joint Assessment and Monitoring Programme (JAMP) on PAH concentrations in sediments and fish bile, and EROD induction, DNA adducts and pre-neoplastic lesions in fish liver, but there is no systematic method for combining this information in order to predict the likelihood of liver neoplasia occurring, either now or in the future. The ACME further noted that in 1999 WGBEC and JBSAEM considered this issue in greater depth and concluded that although expert systems would be ideal for this purpose, they presuppose that the complex system is well understood, and this is not yet the case for the effects of contaminants on the marine environment. However, other types of artificial intelligence (AI) systems may be more appropriate in the present state of partial knowledge.

One possibility is to use self-learning neural networks that can predict particular outcomes on the basis of a large training data set. However, they have the serious disadvantage that they cannot explain in detail how the conclusion is reached. A more promising approach is to use fuzzy logic-based models (as complementing the crisp logic used by experts) trained by a neural network (so-called NeuroFuzzy systems). These use fuzzy sets and rules to take a series of input variables and give an output variable, and they allow the rules to be examined externally. Rules can be generated by experts, by statistical methods such as classification trees, or randomly, and degrees of certainty can be attached to them. They can be 're-tuned' for a different species or area by re-defining the fuzzy variables and rule contributions. The use of fuzzy sets allows words rather than numbers to be used in formulating the rules, e.g., 'if X is high, then Y is low'. This has the benefit of being easier to understand by the experts whose judgement the system is trying to mimic. Again, such systems need large training data sets with which to hone the rules, before they can be considered reliable.

It is considered worthwhile to investigate further the applicability of NeuroFuzzy approaches by trying them out with some appropriate data sets. Sufficiently comprehensive data sets are rare, but it is known that one exists for Puget Sound on the Pacific coast of the USA, where the effects of PAHs on fish have been studied by the U.S. National Oceanic and Atmospheric Administration (NOAA) for many years. It is possible that some smaller data sets generated in the North Sea may also be usable when combined with expert knowledge. The benefit of NeuroFuzzy systems is that they can learn from data in areas where knowledge is weak, and show which knowledge is most reliable in predicting the output variable(s). Once a series of rules has been produced, an iterative process of improvement and assessment of their robustness can begin. The rules provide a strawman for the experts to focus on. This may generate new ideas (rules) that can be fed back into the system. The training data set may also be used to provide subsets of variables in order to test robustness and which factors are the best predictors.

The ACME considers that the feasibility of using NeuroFuzzy techniques for assessing complex marine monitoring data sets should be studied by attempting to train suitable software with a large data set, providing that access to the relevant data can be arranged.

*Need for further research or additional information*

Considerable further research is required to validate the basic NeuroFuzzy approach using PAH-relevant data, and then to assess the problems involved in extrapolating from the training data set to other geographical situations.

The ACME agreed that the development of NeuroFuzzy systems for evaluating complex marine data sets should be commenced through a collaborative workshop between ICES and other interested parties such as NOAA and Pacific ICES (PICES).

### 5.1.4 Suites of biological methods for use in brackish water systems

*Request*

There is no specific request; this is part of continuing ICES work to evaluate marine biological effects monitoring techniques with a view to improving the efficiency of monitoring programmes.

*Source of the information presented*

The 1999 report of the Working Group on Biological Effects of Contaminants (WGBEC) and ACME deliberations.

*Status/background information*

Many of the methods recommended by ACME for monitoring the biological effects of contaminants (see Section 5.1.1, above) have been developed principally for use in fully saline marine waters and not for the brackish waters found in many estuaries and some coastal and offshore areas (especially the Baltic Sea). It is known, however, that the toxicity exerted by some contaminants is different under brackish conditions, partly due to differences in bioavailability, and partly to differences in susceptibility brought about by the greater physiological stress experienced by organisms in areas of low or fluctuating salinity. However, much remains to be clarified in this field, and the various bioassays and biomarkers recommended by ACME should be validated for use under such conditions.

*Need for further research or additional information*

It is clear that considerable further research is required to investigate the responses of the various ACME recommended biological effects monitoring techniques under brackish water conditions. Specifically, it is desirable to know the degree to which low and/or fluctuating salinities increase (or decrease) the sensitivity of the various biomarkers and bioassays currently in use, or proposed for future use, and if necessary to derive correction factors to account for these effects. Experiments must be designed with care to control for changes in the bioavailable fraction of a contaminant during conditions of changing salinity.

The ACME endorses the need to validate biological effects monitoring techniques for use under low or fluctuating salinity conditions.

### 5.1.5 Outcome of the AMAP/EEA/ICES Workshop on Combined Effects in the Marine Environment

*Request*

There is no specific request; this is part of the continuing ICES work on monitoring the biological effects of contaminants.

*Source of the information presented*

The 1999 reports of the Working Group on Biological Effects of Contaminants (WGBEC) and the Working Group on Environmental Assessment and Monitoring Strategies (WGEAMS), and ACME deliberations.

*Status/background information*

In its 1998 report, the ACME emphasized that the interactions between individual, and groups of, contaminants with respect to biological responses remain a difficult and largely unresolved issue in toxicology and marine environmental science and encouraged initiatives to improve knowledge, in view of the importance of this issue (ICES, 1999).

In November 1998, ICES co-sponsored a Workshop on Combined Effects in the Marine Environment, together with the Arctic Monitoring and Assessment Programme (AMAP) and the European Environment Agency (EEA). The objective of the Workshop was to assess methods to detect combined effects of contaminants and either organic enrichment, UV-radiation and/or climate change.

The Workshop organized its work into four areas: (1) reproduction and population studies, (2) cellular and biochemical methods, (3) effects on plankton, pelagic eggs, and pelagic fish larvae, and whole organism bioassays, and (4) pathology and diseases.

At its 1999 meeting, WGBEC reviewed the report of the Workshop and agreed that the topic of combined effects of pollutants is very complex and that it must be realized that most monitoring activities will be dealing with a matrix containing a large number of factors. Various factors will differ in their impact on biological effects measures. Effects may be additive, antagonistic or synergistic. Due to the complexity of the topic, the Workshop did not recommend guidelines for how interactions could be assessed, other than pointing out the importance of keeping relevant factors in mind. In this way, it was envisaged that required knowledge would accumulate. A major conclusion of the Workshop was that there is a need for more basic research. The Workshop also emphasized the importance of using, for example, methods that detect effects on physiological sensory processes and their links to behaviour.

The various techniques discussed at the Workshop will be applied in different geographical areas with a broad range of environmental conditions. The most important environmental factors are temperature, salinity, and light. These factors modulate the response of most biological effects techniques. The Workshop emphasized the need for more knowledge on the use of selected biological methods under differing environmental conditions (e.g., in the Baltic Sea in relation to the North Sea or the Mediterranean Sea).

WGBEC supported the main aim and the results reported from the Workshop; however, it was difficult to obtain a clear understanding of the Workshop's conclusions and recommendations. The recommendations from the four workgroups at the Workshop differed in their relevance. From the report of workgroup 1 (reproduction and population studies), it was noted that chemosensory tests were recommended for use, which was considered an important component. The other recommendations from workgroups 1 and 2 (cellular and biochemical methods) were in line with the opinions expressed by WGBEC and ACME previously. The recommendations from workgroup 3 (plankton, eggs and larvae) were also considered to be scientifically sound and the gaps in knowledge that were identified were correct. There were few members in workgroup 4 (pathology and diseases), but their recommendations were sound, although not always relating to combined effects.

The ACME noted that the outcome of the Workshop is generally seen as a step forward in the knowledge of combined biological effects of contaminants.

*Need for further research or additional data*

The ACME is aware that several studies on the development, evaluation, and application of techniques are under way in the field of combined effects of contaminants. However, due to the complexity and importance of the topic, the ACME encourages researchers to increase their efforts to conduct more applied and experimental research to improve the knowledge on combined biological effects of contaminants.

In this respect, the ACME also recommends that, within the framework of ICES, participation from Canada and the USA in these research activities be encouraged, in order to include their experience in the development of new approaches.

**Reference**

ICES. 1999. Report of the ICES Advisory Committee on the Marine Environment, 1998. ICES Cooperative Research Report, 233: 11–12.

## 5.2 Monitoring Contaminants in Baltic Sea Sediments

*Request*

Item 4 of the 1999 requests from the Helsinki Commission: to arrange, together with HELCOM, a joint Workshop to discuss and present the results of the Baseline Study of Contaminants in Baltic Sea Sediments.

*Source of the information presented*

Version 6.0 of the Report of the 1993 ICES/HELCOM Baseline Study of Contaminants in Baltic Sea Sediments, the report of the ICES/HELCOM Workshop on Baltic Sea Sediments: Conditions and Contaminants (WKBSED), and ACME deliberations.

*Status/background information*

The ACME reviewed the history of the joint ICES/HELCOM work to study sediments in the Baltic Sea, primarily based on the 1993 Baseline Study of Contaminants in Baltic Sea Sediments and, more recently, on the report of the ICES/HELCOM Workshop on Baltic Sea Sediments: Conditions and Contaminants, held in April 1999. The ACME noted that it has been difficult to provide overall conclusions and recommendations based on these activities, at least partly owing to the diversity of conditions in the various areas of the Baltic Sea. The results of the Baseline Study have shown that the hydrological conditions in the various basins of the Baltic Sea are very dynamic and this is reflected in the history of sedimentation in these basins. Therefore, the decision on the selection of the basins for further sediment studies is not easy and should be supported by additional measurements in each of the candidate basins, including a thorough geological survey. While the results of the Baseline Study show that some Baltic basins may be useful for monitoring, not all basins are suitable for monitoring based on the use of sediment cores. Thus, the ACME is of the opinion that only some carefully selected Baltic basins, where relatively undisturbed sedimentation occurs, can be recommended for sediment monitoring based on the analysis of sediment cores.

A new baseline study could be conducted ten years after the previous one, i.e., in 2003, as a time interval of five to ten years may be desirable for identifying changes in Baltic sediments. However, this new exercise should be based on detailed planning prepared with clear objectives for the conduct of the study, a clearly planned programme of sampling and analysis, and an agreed plan for the submission and evaluation of the data.

*Recommendations*

ICES recommends that the results of the Baseline Study of Contaminants in Baltic Sea Sediments be made available to the ICES data bank and the scientific community.

ICES also recommends that the results of the 1993 Baseline Study, together with other more recent results, be used in the development of a 'sediment chapter' for the Fourth Periodic Assessment of the State of the Marine Environment of the Baltic Sea.

ICES recommends that the planning of the future monitoring of sediments in the Baltic Sea should be carried out by an appropriate body under HELCOM based on the experience and information gained during the1993 Baseline Study.

## 5.3 Arctic Monitoring and Assessment Programme: Phase II Programme

*Request*

There is no request; this is ongoing work in cooperation with the Arctic Monitoring and Assessment Programme (AMAP).

*Source of the information presented*

Various AMAP reports and ACME deliberations.

*Status/background information*

The ACME was informed that the Arctic Monitoring and Assessment Programme (AMAP) at its twelfth Working Group meeting, which took place in Helsinki, Finland in December 1998, approved a new work plan for the next five years. The new plan combines targeted monitoring of hot spots and long-term trends, specific clean-up projects in the Russian Arctic for PCBs and mercury, and new cooperative monitoring projects on UV-radiation and climate variation with the Conservation of Arctic Flora and Fauna (CAFF) and the International Arctic Science Committee (IASC).

The work in the new phase of AMAP will be coordinated by an Assessment Steering Group (ASG-II), which consists of experts on human health, heavy metals, persistent organic pollutants (POPs), radioactivity, acidification, oil, and combined effects. The ASG-II may also take over many of the current functions of AMAP's Working Group. In addition, a separate Assessment Steering Committee will coordinate the work on UV and climate change with CAFF.

The AMAP programme for gathering new environmental data, including information on pollution and human health, has been under preparation during 1998 and 1999. The overall monitoring programme has been divided into a levels and trends programme and an effects programme, with an additional part addressing the needs for source-related information. Individual components within these programmes specify priority contaminants to be monitored, methodologies to be applied, quality assurance and quality control procedures, and other related issues such as the flow of data. The ICES Environmental Data Centre serves as the Thematic Data Centre for the marine component of AMAP data. The draft sub-programmes for the AMAP monitoring programme developed by expert groups used different styles, formats, and concepts and, consequently, considerable work is currently under way to prepare a consistent documentation of the programme for the coming period. The final programme plan is expected to be circulated in summer 1999.

As part of the cooperation between AMAP, the European Environment Agency (EEA), and ICES, an international Workshop on Combined Effects in the Marine Environment was held in Copenhagen on 16–17 November 1998 (see Section 5.1.5, above). The Workshop report recommending methodologies for related scientific work and future cooperation involving research and field experiments has been produced and circulated. Combined effects will be a very important issue for AMAP during Phase II.

Parts of the AMAP programme concerning climate and UV effects issues are under development in close cooperation with CAFF, IASC, and other international programmes involved in such studies. For the marine climate component, the observation network operated by ICES Member Countries will serve as an important source of information. It is intended that the programme plan will be finalized during the autumn of 1999.

During Phase II of AMAP, individual assessment reports for the different components will be prepared instead of a joint report, as was prepared for Phase I (AMAP, 1998a). Three types of reports are planned: progress reports, interim reports, and main reports. The progress reports will give information on progress in the preparation of an assessment report for inclusion in the AMAP Progress Report to the Ministerial Meeting of the Arctic Council, while the interim reports will give intermediate information on the results obtained after the last assessment, including the major results reported at the conferences and symposia organized under AMAP auspices. The main reports will include detailed information on the results obtained by AMAP, in cooperation with the other fora, according to the requests of the Ministers, including recommendations for actions.

For more information, see also AMAP on the Internet at http://www.grida.no/amap/.

The timetable for the AMAP assessment reports is as follows:

| Assessment item | Year of reporting | | | |
|---|---|---|---|---|
| | Adopted | | Tentative | |
| | 2000 | 2002 | 2004 | 2006 |
| Human health | P | M | I | M |
| POPs | P | M | I | M |
| Heavy metals | P | M | I | M |
| Radioactivity | P | M | I | M |
| Acidification | P | P | P | M |
| Oil and PAHs | P | P | M | P |
| TBT | P | P | M | P |
| Climate effects | I | M | P | M |
| UV-B effects | I | M | P | M |
| Combined effects | P | P | M | P |

P = progress report; I = interim report; M = main report

**References**

AMAP. 1998a. AMAP Assessment Report: Arctic pollution issues. AMAP, Oslo, Norway. 859 pp.

AMAP. 1998b. Minutes of the Twelfth Meeting of the Arctic Monitoring and Assessment Programme Working Group (AMAPWG), 7–9 December 1998, Helsinki, Finland.

AMAP. 1999a. Minutes of the First Meeting of the Assessment Steering Group (ASG-II) of the Arctic Monitoring and Assessment Programme, 8–10 March 1999, Silver Springs, MD, USA.

AMAP. 1999b. Minutes of the First Meeting of the Assessment Steering Committee, 11 March 1999, Silver Springs, MD, USA.

AMAP. 1999c. Progress Report from AMAP to the SAO meeting, 5–6 May 1999, Anchorage, Alaska, USA.

WWF. 1999. Arctic Bulletin, No 1.

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

*Request*

There is no specific request; this updates information presented in previous ACME reports and is of interest to organizations coordinating international or regional monitoring programmes on nutrients and contaminants in marine media.

*Source of the information presented*

The 1999 report of the Marine Chemistry Working Group (MCWG) and ACME deliberations.

*Status/background information*

The overall performance within the QUASIMEME Laboratory Performance Scheme (LPS) for a given parameter has been used as an indicator of the ability of laboratories to perform routine monitoring. Tabular presentations have been prepared that outline the performance of laboratories as a whole group. These tables provide a summary of results from ten or eleven exercises carried out over a period of 2.5 years (June 1996–December 1998) for determinands contained in the QUASIMEME II programme (CBs, organochlorine pesticides (OCPs), and trace elements in biota; trace elements, CBs, OCPs, and PAHs in sediments; and nutrients in sea water). It must be realized that the results from all participating laboratories are included, and not specifically OSPAR laboratories or groups of laboratories representing the ICES community, for which comparable material is not presently available.

In the QUASIMEME II scheme, analytical performance is evaluated using individual laboratory Z scores. For a particular contaminant/medium combination in an intercomparison exercise, these are defined as

$$Z = \frac{\bar{c} - \bar{\bar{c}}}{k}$$

where $\bar{c}$ is the laboratory mean contaminant concentration, $\bar{\bar{c}}$ is an assigned value, usually given by the mean for a group of reference laboratories, and $k$ is an externally defined total allowable error for laboratory bias, taking a value of 12.5 % (6 % for nutrients) and increasing to 50 % towards the limit of detection.

A global target $|Z| < 2$ is used to characterize 'satisfactory' analytical performance by a laboratory. The criterion for satisfactory group performance for a contaminant/medium combination is that at least 75 % of the laboratories have attained a satisfactory analytical performance.

The ACME noted that this definition of 'satisfactory' provides a reasonable common criterion for summarizing and comparing performance. However, it should not be used as a necessary and sufficient condition for ensuring that monitoring data are adequate for monitoring purposes. Appropriate targets for analytical performance will depend on the monitoring objectives, the sampling scheme, and the treatment of samples.

QUASIMEME results for the determination of chlorobiphenyls (CBs) and organochlorine pesticides (OCPs) are presented in Table 5.4.1 for biological tissues and in Table 5.4.2 for sediments. Table 5.4.3 shows the QUASIMEME results for the determination of polycyclic aromatic hydrocarbons (PAHs) in sediments.

With regard to trace elements, the results of QUASIMEME exercises for analysis of biological tissues and for analysis of sediments are shown in Table 5.4.4 and Table 5.4.5, respectively. Finally, the results of QUASIMEME exercises on the analysis of nutrients in marine and estuarine waters are listed in Table 5.4.6.

The results from the nutrient exercises merit some specific comments. In the period mentioned above, a total of ten intercomparison samples for oceanic waters and eleven samples for estuarine waters, covering a range of concentrations, had been distributed for the analysis of dissolved ammonia, nitrite, phosphate, silicate, total nitrogen, total phosphorus, and total oxidized nitrogen (nitrate + nitrite) (designated TOxN in the table). For either type of water, up to 40–50 laboratories had returned results. The overall assessment of these groups of laboratories can be taken as an indication of their capacity to monitor nutrients. The group success indicator shows the number of intercomparison rounds for which the performance of the group as a whole was satisfactory. When comparing the laboratories' results from 1997 and 1998, it appears that some determinands that did not give rise to a fully satisfactory performance in 1997 actually met satisfactory performance requirements in 1998. These determinands are ammonia and phosphate. Some improvement was also shown for Total-N. For Total-P, some laboratories still encounter problems and the overall group performance is not entirely satisfactory.

With respect to trace elements, Table 5.4.7 provides recent information on the minimum trace metal concentrations in sediments and biota for which more than 60 % of the laboratories achieved $|Z|$ scores $< 2$.

The tables presented here give an idea of the overall performance of the laboratories as a whole group. From the tables, it is clear that not all laboratories are able to meet the criterion of $|Z| < 2$ for the different parameters, particularly for OCPs. The laboratory performance with respect to the nutrients can, however, be regarded as more encouraging, as it must be realized that a situation in which all laboratories simultaneously show good performance is unlikely to materialize.

These tables also show that laboratories used in international marine monitoring programmes should be carefully selected and that they should always submit quality assurance information together with their monitoring data. It is important to keep track of the long-term performance of the laboratories, and it is urged that long-term proficiency testing schemes be followed.

*Need for further research or additional data*

The ACME noted that MCWG intends to update these tables each year, using information on laboratory performance only from the three most recent years.

**Table 5.4.1.** Summary assessment of laboratory group performance in QUASIMEME exercises on the determination of chlorobiphenyls and organochlorine pesticides in biological tissues, June 1996–December 1998.

| Determinand | Units | [1]Range of assigned values | [2]Range of ±target bias (%) | [3]Range of between lab CVs (%) | [4]Range for % No. obs. with $\|Z\| < 2$ | [5]Satisfactory group performances/total rounds |
|---|---|---|---|---|---|---|
| CB028 | $\mu g\ kg^{-1}$ | 0.31–13.63 | 13–29 | 33.6–89.7 | 41.7–79.3 | 3/10 |
| CB052 | $\mu g\ kg^{-1}$ | 0.53–34.68 | 13–22 | 28.3–68.2 | 40.0–76.7 | 2/10 |
| CB101 | $\mu g\ kg^{-1}$ | 1.47–106.75 | 13–16 | 17.5–55.0 | 45.2–78.1 | 3/10 |
| CB105 | $\mu g\ kg^{-1}$ | 0.38–40.79 | 13–26 | 27.1–58.3 | 44.0–75.0 | 1/10 |
| CB118 | $\mu g\ kg^{-1}$ | 1.05–147.29 | 13–17 | 21.0–39.0 | 51.7–81.3 | 2/10 |
| CB138 | $\mu g\ kg^{-1}$ | 2.32–286.91 | 13–15 | 23.8–35.6 | 48.3–79.3 | 2/10 |
| CB156 | $\mu g\ kg^{-1}$ | 0.15–16.60 | 13–45 | 27.0–82.4 | 53.3–84.0 | 6/10 |
| CB180 | $\mu g\ kg^{-1}$ | 0.43–90.03 | 13–24 | 22.8–45.2 | 51.9–87.5 | 7/10 |
| HCB | $\mu g\ kg^{-1}$ | 0.07–19.53 | 13–82 | 30.4–67.8 | 48.1–80.6 | 5/10 |
| *p,p'*-DDE | $\mu g\ kg^{-1}$ | 0.58–198.02 | 13–21 | 20.9–56.8 | 51.9–75.0 | 3/10 |
| β-HCH | $\mu g\ kg^{-1}$ | 0.07–2.52 | 14–81 | 0.0–93.4 | 26.7–47.4 | 3/6 |
| γ-HCH | $\mu g\ kg^{-1}$ | 0.07–3.07 | 14–89 | 48.6–121.6 | 48.1–73.3 | 4/10 |
| *p,p'*-DDD | $\mu g\ kg^{-1}$ | 0.25–53.59 | 13–32 | 22.2–82.2 | 45.8–75.0 | 4/10 |
| *p,p'*-DDT | $\mu g\ kg^{-1}$ | 0.29–26.28 | 13–30 | 50.4–122.4 | 0.0–55.6 | 0/10 |
| *o,p*-DDT | $\mu g\ kg^{-1}$ | 0.09–17.71 | 13–68 | 98.7–160.7 | 37.5–52.9 | 0/9 |
| Dieldrin | $\mu g\ kg^{-1}$ | 0.53–50.85 | 13–22 | 31.8–66.5 | 18.8–83.3 | 1/10 |
| *Trans*-nonachlor | $\mu g\ kg^{-1}$ | 0.12–22.27 | 13–55 | 26.3–94.0 | 64.7–100.0 | 7/10 |
| Lipid, extracted | % | 0.94–56.15 | 13–18 | 5.6–35.5 | 69.2–100.0 | 7/8 |
| Lipid, total | % | 1.11–58.34 | 10–26 | 4.5–40.2 | 0.0–100.0 | 7/8 |

**Table 5.4.2.** Summary assessment of laboratory group performance in QUASIMEME exercises on the determination of chlorobiphenyls and organochlorine pesticides in sediments, June 1996–December 1998.

| Determinand | Units | [1]Range of assigned values | [2]Range of ±target bias (%) | [3]Range of between lab CVs (%) | [4]Range for % No. obs. with $\|Z\| < 2$ | [5]Satisfactory group performances/total rounds |
|---|---|---|---|---|---|---|
| CB028 | $\mu g\ kg^{-1}$ | 0.15–34.08 | 17–46 | 20.0–52.3 | 39.1–94.4 | 7/10 |
| CB052 | $\mu g\ kg^{-1}$ | 0.19–28.36 | 17–47 | 28.4–72.7 | 52.9–88.2 | 6/10 |
| CB101 | $\mu g\ kg^{-1}$ | 0.50–33.59 | 23–48 | 17.3–47.3 | 60.0–85.7 | 5/10 |
| CB105 | $\mu g\ kg^{-1}$ | 0.14–5.76 | 14–36 | 25.4–111.7 | 42.9–100.0 | 6/10 |
| CB118 | $\mu g\ kg^{-1}$ | 0.36–25.54 | 20–47 | 22.6–58.3 | 52.9–78.3 | 7/10 |
| CB138 | $\mu g\ kg^{-1}$ | 1.03–37.93 | 24–47 | 25.1–42.7 | 41.2–83.3 | 4/10 |
| CB153 | $\mu g\ kg^{-1}$ | 0.94–45.34 | 25–48 | 18.1–44.4 | 52.0–82.6 | 3/10 |
| CB156 | $\mu g\ kg^{-1}$ | 0.10–3.67 | 16–33 | 27.9–71.7 | 43.8–81.3 | 7/10 |
| CB180 | $\mu g\ kg^{-1}$ | 0.59–26.68 | 24–48 | 19.5–59.8 | 41.7–75.0 | 2/10 |
| HCB | $\mu g\ kg^{-1}$ | 0.12–13.92 | 16–41 | 23.7–57.8 | 40.0–88.9 | 5/10 |
| *p,p'*-DDE | $\mu g\ kg^{-1}$ | 0.40–6.77 | 20–41 | 14.3–65.3 | 46.2–88.9 | 6/10 |
| α-HCH | $\mu g\ kg^{-1}$ | 0.08–0.41 | 11–34 | 43.1–97.1 | 0.0–93.3 | 8/10 |
| γ-HCH | $\mu g\ kg^{-1}$ | 0.14–0.58 | 11–32 | 39.6–118.9 | 46.7–83.3 | 6/10 |
| *p,p'*-DDD | $\mu g\ kg^{-1}$ | 0.49–12.00 | 16–38 | 34.8–60.4 | 0.0–66.7 | 0/10 |
| *p,p'*-DDT | $\mu g\ kg^{-1}$ | 0.25–4.40 | 11–33 | 44.5–124.3 | 0.0–73.3 | 0/10 |
| *o,p*-DDT | $\mu g\ kg^{-1}$ | 0.08–0.58 | 4–16 | 40.6–134.7 | 0.0–88.9 | 6/10 |
| Dieldrin | $\mu g\ kg^{-1}$ | 0.17–1.69 | 9–24 | 37.5–86.2 | 50.0–90.0 | 3/10 |
| *Trans*-nonachlor | $\mu g\ kg^{-1}$ | 0.05–0.42 | 6–12 | 60.5–114.7 | 50.0–50.0 | 8/10 |
| TOC | % | 2.42–4.20 | 6–7 | 9.1–26.1 | 58.8–100.0 | 4/4 |

[1]Range of assigned values for all rounds of the QUASIMEME scheme examined. The determined assigned values are only indicative.
[2]Target bias or total allowable error. This is calculated as: total error % = fixed error (12.5) % + (constant error/concentration) %. Thus, the total error is dependent on the concentration of the determinand.
[3]Range of between-laboratory coefficients of variance (CVs) (%) over all rounds examined.
[4]Range of the number of laboratories achieving the set QUASIMEME standard of $|Z| < 2$ (expressed as %).
[5]Number of rounds in which an overall satisfactory performance has been achieved, expressed as a fraction of the total number of rounds for which total assigned values could be derived. Performance is considered satisfactory when the robust CV % – (total error × 2) > 0.

**Table 5.4.3.** Summary assessment of laboratory group performance in QUASIMEME exercises on the determination of PAHs in sediments, June 1996–December 1998.

| Determinand | Units | [1]Range of assigned values | [2]Range of ±target bias (%) | [3]Range of between lab CVs (%) | [4]Range for % No. obs. with \|Z\| < 2 | [5]Satisfactory group performances/total rounds |
|---|---|---|---|---|---|---|
| Acenaphthene | mg $kg^{-1}$ | 0.01–0.43 | 12–17 | 29.6–95.0 | 29.4–89.5 | 2/8 |
| Anthracene | mg $kg^{-1}$ | 0.03–0.43 | 21–21 | 29.6–41.4 | 45.0–88.0 | 1/4 |
| Benzo[*a*]anthracene | mg $kg^{-1}$ | 0.08–1.18 | 25–41 | 15.9–47.4 | 56.0–79.2 | 3/10 |
| Benzo[*a*]pyrene | mg $kg^{-1}$ | 0.09–1.17 | 25–41 | 22.2–36.8 | 42.9–77.3 | 2/10 |
| Benzo[*b*]fluoranthene | mg $kg^{-1}$ | 0.13–1.42 | 24–41 | 28.2–56.0 | 44.0–68.2 | 0/10 |
| Benzo[*e*]pyrene | mg $kg^{-1}$ | 0.10–1.34 | 19–32 | 15.8–43.5 | 41.2–87.5 | 6/10 |
| Benzo[*ghi*]perylene | mg $kg^{-1}$ | 0.10–1.22 | 25–37 | 12.8–54.2 | 52.4–87.5 | 3/10 |
| Benzo[*k*]fluoranthene | mg $kg^{-1}$ | 0.07–0.57 | 20–20 | 32.9–52.1 | 20.0–68.2 | 0/4 |
| Chrysene | mg $kg^{-1}$ | 0.10–1.48 | 25–41 | 18.9–46.7 | 36.4–75.0 | 2/10 |
| Dibenz[*ah*]anthracene | mg $kg^{-1}$ | 0.02–0.13 | 19–20 | 49.0–70.4 | 33.3–81.8 | 1/4 |
| Fluoranthene | mg $kg^{-1}$ | 0.18–2.46 | 24–41 | 16.8–34.2 | 60.0–81.8 | 6/10 |
| Fluorene | mg $kg^{-1}$ | 0.03–0.35 | 17–17 | 39.5–61.9 | 52.4–85.7 | 1/4 |
| Indeno[1,2,3-*cd*]pyrene | mg $kg^{-1}$ | 0.10–1.16 | 23–37 | 27.3–48.3 | 43.5–72.0 | 2/10 |
| Naphthalene | mg $kg^{-1}$ | 0.06–0.87 | 18–19 | 51.8–84.4 | 50.0–100.0 | 0/4 |
| Phenanthrene | mg $kg^{-1}$ | 0.14–1.44 | 24–41 | 20.1–35.4 | 56.0–80.0 | 5/10 |
| Pyrene | mg $kg^{-1}$ | 0.14–2.16 | 25–41 | 14.2–30.1 | 68.0–83.3 | 5/10 |
| TOC | % | 2.73–3.71 | 4–5 | 11.0–13.8 | 66.7–100.0 | 4/4 |

**Table 5.4.4.** Summary assessment of laboratory group performance in QUASIMEME exercises on the analysis of trace elements in biological tissues, June 1996–December 1998.

| Determinand | Units | [1]Range of assigned values | [2]Range of ±target bias (%) | [3]Range of between lab CVs (%) | [4]Range for % No. obs. with \|Z\| < 2 | [5]Satisfactory group performances/total rounds |
|---|---|---|---|---|---|---|
| Arsenic | µg $kg^{-1}$ | 1300–6905 | 14–21 | 18.9–47.4 | 55.0–95.0 | 9/11 |
| Cadmium | µg $kg^{-1}$ | 5.15–458 | 13–187 | 13.5–95.4 | 39.3–80.0 | 6/11 |
| Chromium | mg $kg^{-1}$ | 0.10–2.31 | 13–49 | 49.6–112.0 | 42.1–95.7 | 1/11 |
| Copper | mg $kg^{-1}$ | 0.26–6.27 | 13–31 | 12.1–34.1 | 26.3–91.4 | 9/11 |
| Dry weight | % | 13.97–71.22 | 13–13 | 2.6–12.3 | 73.7–94.1 | 11/11 |
| Lead | µg $kg^{-1}$ | 14.24–892 | 13–31 | 40.0–148.9 | 35.0–100.0 | 0/11 |
| Mercury | µg $kg^{-1}$ | 27.87–122 | 17–45 | 15.1–40.6 | 71.0–90.9 | 11/11 |
| Nickel | mg $kg^{-1}$ | 0.05–1.44 | 14–70 | 37.2–107.4 | 53.8–81.8 | 6/11 |
| Selenium | µg $kg^{-1}$ | 396–1258 | 13–14 | 16.3–38.0 | 58.3–100.0 | 2/7 |
| Zinc | mg $kg^{-1}$ | 4.47–25.95 | 13–28 | 9.7–20.4 | 74.3–100.0 | 11/11 |

[1]Range of assigned values for all rounds of the QUASIMEME scheme examined. The determined assigned values are only indicative.
[2]Target bias or total allowable error. This is calculated as: total error % = fixed error (12.5) % + (constant error/concentration) %.
Thus, the total error is dependent on the concentration of the determinand.
[3]Range of between-laboratory coefficients of variance (CVs) (%) over all rounds examined.
[4]Range of the number of laboratories achieving the set QUASIMEME standard of |Z| < 2 (expressed as %).
[5]Number of rounds in which an overall satisfactory performance has been achieved, expressed as a fraction of the total number of rounds for which total assigned values could be derived. Performance is considered satisfactory when the robust CV % – (total error × 2) > 0.

**Table 5.4.5.** Summary assessment of laboratory group performance in QUASIMEME exercises on the analysis of trace elements in sediments, June 1996–December 1998.

| Determinand | Units | [1]Range of assigned values | [2]Range of ±target bias (%) | [3]Range of between lab CVs (%) | [4]Range for % No. obs. with $\lvert Z\rvert < 2$ | [5]Satisfactory group performances/total rounds |
|---|---|---|---|---|---|---|
| Aluminium § | % | 1.24–6.41 | 26–62 | 22.9–54.6 | 42.4–96.4 | 1/10 |
| Arsenic | $mg\ kg^{-1}$ | 2.21–23.89 | 26–63 | 12.2–42.9 | 48.1–96.2 | 9/10 |
| Cadmium | $\mu g\ kg^{-1}$ | 23.14–10977 | 30–70 | 13.0–71.8 | 40.0–84.8 | 6/10 |
| Chromium | $mg\ kg^{-1}$ | 28.62–323 | 24–66 | 11.9–40.1 | 63.6–100.0 | 9/10 |
| Chromium-P* | $mg\ kg^{-1}$ | 45.77–136 | 11–23 | 20.0–32.3 | 54.5–84.6 | 1/4 |
| Copper | $mg\ kg^{-1}$ | 1.54–189 | 37–79 | 7.3–64.9 | 45.9–100.0 | 9/10 |
| Iron | % | 2.78–4.71 | 32–63 | 6.6–12.1 | 78.0–93.8 | 6/6 |
| Lead | $mg\ kg^{-1}$ | 7.82–233 | 37–79 | 8.6–44.7 | 44.4–95.3 | 9/10 |
| Lithium | $mg\ kg^{-1}$ | 35.70–66.38 | 20–34 | 10.6–16.8 | 88.2–97.1 | 6/6 |
| Manganese | $mg\ kg^{-1}$ | 747–1130 | 32–59 | 7.3–11.1 | 83.3–100.0 | 6/6 |
| Mercury | $\mu g\ kg^{-1}$ | 6.09–2237 | 24–69 | 10.7–76.9 | 73.3–95.1 | 9/10 |
| Nickel | $mg\ kg^{-1}$ | 3.07–60.58 | 34–76 | 9.6–54.8 | 29.4–100.0 | 9/10 |
| Scandium | $mg\ kg^{-1}$ | 7.61–11.28 | 4–10 | 6.4–34.3 | 60.0–100.0 | 5/6 |
| TOC | % | 0.06–6.24 | 11–33 | 13.9–97.9 | 72.7–90.9 | 10/10 |
| Zinc | $mg\ kg^{-1}$ | 10.11–1051 | 37–79 | 5.6–45.6 | 40.0–100.0 | 9/10 |

§Based on both total and partial methods. It is recommended that only values based on total digestion methods should be used in the future.
*P means partial dissolution without hydrofluoric acid.

**Table 5.4.6.** Summary assessment of laboratory group performance in QUASIMEME nutrient exercises, June 1996–December 1998.

| Determinand | Units | [1]Range of assigned values | [2]Range of ±target bias (%) | [3]Range of between lab CVs (%) | [4]Range for % No. obs. with $\lvert Z\rvert < 2$ | [5]Satisfactory group performances/total rounds |
|---|---|---|---|---|---|---|
| **Nutrients in Sea Water** | | | | | | |
| Ammonia | $\mu mol\ l^{-1}$ | 0.90–22.10 | 7–34 | 12.8–64.4 | 33.9–92.2 | 8/10 |
| Nitrite | $\mu mol\ l^{-1}$ | 0.42–1.77 | 7–12 | 3.8–15.9 | 80.0–92.2 | 10/10 |
| Phosphate | $\mu mol\ l^{-1}$ | 0.05–1.65 | 8–61 | 5.3–76.3 | 66.7–94.1 | 9/10 |
| Silicate | $\mu mol\ l^{-1}$ | 1.79–17.20 | 7–12 | 6.7–20.9 | 65.2–89.4 | 10/10 |
| Total-N | $\mu mol\ l^{-1}$ | 8.76–51.60 | 6–9 | 7.3–23.1 | 52.9–84.6 | 7/10 |
| Total-P | $\mu mol\ l^{-1}$ | 0.20–1.78 | 7–19 | 6.7–43.9 | 62.5–92.3 | 6/10 |
| TOxN | $\mu mol\ l^{-1}$ | 1.19–22.41 | 7–27 | 3.1–16.2 | 83.3–94.5 | 10/10 |
| **Nutrients in Estuarine Water** | | | | | | |
| Ammonia | $\mu mol\ l^{-1}$ | 1.23–24.29 | 7–26 | 9.5–23.0 | 66.7–97.4 | 10/11 |
| Nitrite | $\mu mol\ l^{-1}$ | 0.53–6.37 | 6–11 | 3.7–8.0 | 83.3–97.4 | 11/11 |
| Phosphate | $\mu mol\ l^{-1}$ | 1.15–6.54 | 6–8 | 4.1–15.9 | 62.2–95.1 | 10/11 |
| Silicate | $\mu mol\ l^{-1}$ | 3.26–21.86 | 6–9 | 6.7–10.1 | 78.4–91.1 | 11/11 |
| Total-N | $\mu mol\ l^{-1}$ | 5.73–65.05 | 6–10 | 7.5–21.9 | 57.1–88.2 | 9/11 |
| Total-P | $\mu mol\ l^{-1}$ | 1.28–6.62 | 6–8 | 5.6–21.2 | 47.6–91.7 | 8/11 |
| TOxN | $\mu mol\ l^{-1}$ | 2.64–36.27 | 7–15 | 3.6–8.3 | 81.4–95.1 | 11/11 |

[1]Range of assigned values for all rounds of the QUASIMEME scheme examined. The determined assigned values are only indicative.
[2]Target bias or total allowable error. This is calculated as: total error % = fixed error (12.5) % + (constant error/concentration) %. Thus, the total error is dependent on the concentration of the determinand.
[3]Range of between-laboratory coefficients of variance (CVs) (%) over all rounds examined.
[4]Range of the number of laboratories achieving the set QUASIMEME standard of $\lvert Z\rvert < 2$ (expressed as %).
[5]Number of rounds in which an overall satisfactory performance has been achieved, expressed as a fraction of the total number of rounds for which total assigned values could be derived. Performance is considered satisfactory when the robust CV % – (total error × 2) > 0.

**Table 5.4.7.** Lowest concentrations of trace elements in sediments and biota which can be monitored on a routine basis by the majority of laboratories (outcome of ten QUASIMEME LPS carried out in June 1996–December 1998).

| Trace element | Sediments (dry weight) | Biota (wet weight) |
|---|---|---|
| Zn | 75 mg $kg^{-1}$ | ≤ 4.6 mg $kg^{-1}$ |
| Cd | 340 µg $kg^{-1}$ | For fish tissue, 5 µg $kg^{-1}$ |
| Pb | 40 mg $kg^{-1}$ | Problems for the majority of the labs, even at 1 mg $kg^{-1}$ |
| Cu | 17 mg $kg^{-1}$ | ≤ 0.3 mg $kg^{-1}$ |
| Cr | 28 mg $kg^{-1}$ | Problems for the majority of the labs, even at 2 mg $kg^{-1}$ |
| Ni | 23 mg $kg^{-1}$ | For cod liver and muscle, 0.1 mg $kg^{-1}$ |
| As | 6 mg $kg^{-1}$ | ≤ 1.3 mg $kg^{-1}$ |
| Hg | 120 µg $kg^{-1}$ | ≤ 28 µg $kg^{-1}$ |
| Al | Value not available* | |
| Mn | ≤ 750 mg $kg^{-1}$ | |
| Fe | ≤ 2.8 % | |
| Li | ≤ 35 mg $kg^{-1}$ | |
| Sc | ≤ 7.6 mg $kg^{-1}$ | |

'≤' means that only a less than concentration can be given and not a minimum concentration for which the majority of the laboratories is able to analyse. A minimum concentration could not be calculated from the results of the QUASIMEME LPS, as the concentrations of these analytes in the samples used were not low enough.

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

Additionally, an effort will be made to prepare a similar table in 2000 based on a selection of laboratories, particularly those reporting data to ICES and OSPAR.

## 5.5 Techniques for Sediment Monitoring: Normalization

*Request*

There is no specific request; this is part of the continuing ICES work on techniques for sediment monitoring and is related to previous requests from the OSPAR Commission.

*Source of the information presented*

The 1999 report of the Working Group on Marine Sediments in Relation to Pollution (WGMS) and ACME deliberations.

*Status/background information*

The ACME noted that in 1999 WGMS continued its consideration of the topic of normalization techniques for contaminant concentrations in sediments. In this connection, WGMS reviewed and evaluated relevant information from the EC Standards, Measurements and Testing Programme-funded project 'Quality Assurance of Sample Handling' (QUASH), due to its importance to the ongoing discussion on normalization.

The QUASH project has the objective of assessing the contribution to the total variability of methods for sampling and sample handling currently being used by laboratories involved in European monitoring programmes. It undertakes to identify factors which are critical to the quality of data, and to evaluate procedures that may be used to overcome problems. Special attention is being paid to the determination of cofactors such as total organic carbon (TOC) in sediments, as such determinands play an important role in the interpretation of the data. Furthermore, the interlaboratory variability of the sieving process has been studied and it was found that the variability of the sieving process (sieving error) was small compared to the interlaboratory analytical variability (analytical error).

WGMS noted results of investigations, that preceded the QUASH exercises, on sediment samples from different estuaries in Europe, where the sieving process was evaluated by analysing the clay content in the remaining coarse fraction, as well as in the fine fraction obtained. The clay content in the fraction < 63 µm showed a range of a factor of four, implying that sieving does not result in physically equal samples. Further grain-size correction appeared to be indispensable. The study also showed that, although the clay content that adhered to the sand particles was small, this could, in the case of sandy samples, still be a considerable amount for coarse-grained samples (< 1 % fines). Using ultrasonic treatment improves the yield, depending on the intensity (i.e., distance of the sieve above the ultrasonic bath bottom).

WGMS reviewed the outcome of relevant QUASH work during 1998. Several key issues were considered, including:

- the need to improve the sieving procedure to make it less laborious;
- the requirement for an evaluation of potential cofactors to determine whether they are the correct ones and to determine which cofactors are best for application to a larger geographical area.

With regard to the first issue, two sieving procedures were reviewed, one applied by the National Institute for Coastal and Marine Management (RIKZ, the Netherlands), and the other by the Institute of Hydrology (BFG, Koblenz, Germany). These procedures are being tested in the framework of QUASH.

The RIKZ technique uses an automatic wet sieving system clamped onto a heavy vibrating table. Sea water is circulated through the sieve by a peristaltic pump. A continuous-flow centrifuge equipped with a titanium rotor (17 000 rpm) is used to separate the sieved particles from the elutriate passing through the sieve. From the centrifuge, the water is returned to the sieve. This closed recycling system reduces the overall water-to-solid ratio and hence any leaching or contamination effects during wet sieving. Sieving time is also considerably reduced with this procedure; large amounts of sediment samples can be processed in a short time. The only drawback of this system is its high cost.

The BFG applies a manual wet sieving batch procedure effected by simultaneous ultrasonic treatment and addition of agate balls for separating the fine fractions < 63 µm or < 20 µm. For subsequent analyses of organic contaminants, organic matter still adhering to coarser fractions is extracted by an additional sedimentation step and added to the fine fraction in order to maximize contaminant yield. De-ionized water is used throughout all steps. Particles and water are separated by batch centrifugation. The system is inexpensive, but time consuming, in particular for sandy sediments and when large amounts of sediment samples need to be processed.

In the QUASH sieving intercomparison exercise, most participants used some variation of the BFG method. It was found that the time needed for the isolation of the material from the sieved suspension by centrifugation will most likely be reduced if saline water is used. Comparison of the yield for the QUASH sediments in this exercise with and without ultrasonic treatment did not reveal a significant difference. However, malfunctioning of the sieving process by an unnoticed hole in the nylon sieving cloth may occur. This will effectively dilute the sample with large particles.

Furthermore, the concept of the next round of QUASH, currently being conducted, was considered by WGMS. Sieving reduces the compositional variation, but further correction for grain-size differences is possible using cofactors. In other words, it is considered that sieving is not an end to normalization, but mainly a concentration of cofactors and contaminants. To validate any correction procedure, the relationship between cofactors and contaminants then should be examined and evaluated using statistical methods such as regression analysis. To perform such an evaluation, a set of samples would be required with different grain-size distributions but the same contamination level, that is, equilibrium at the particle level (or simple cofactor/contaminant relationship). In areas where deposited sediments are composed of material of differing origin, different levels of contamination may occur in different samples, and a relation between contaminant and cofactors should not be expected. It is assumed that equilibrated contamination only occurs in areas without direct inputs and where the sediments are thoroughly mixed by hydrodynamic forces. Such homogeneous areas with a wide range of grain sizes are rare; they may have to be created in the laboratory. In the second QUASH round, the sediment samples are mixed for seven days in the presence of water from the sample location. The sample is then fractionated using 20 µm and 63 µm sieves (i.e., into < 20 µm, < 63 µm, 20–63 µm, > 63 µm fractions, and total samples). As QUASH participants will process samples from their respective regions, it is expected that information will be obtained on how the contaminants and cofactors studied are distributed over size fractions in samples from different geographical regions within Europe. To a certain degree, the effectiveness of cofactors and grain-size correction procedures can be tested by the data obtained. In the next round, this will be extended to a wide geographical area with the aim of finding a suitable cofactor that applies equally well to all areas, or at least an overall acceptable best choice.

As a contribution to the discussion on normalization in addition to the QUASH issue, work conducted in the Netherlands to geochemically characterize sediments in the Dutch continental shelf area was reviewed by WGMS. Pliocene, Pleistocene, and Holocene sub-surface sediments were analysed to provide background values against which other samples could be compared. A system of evaluating the data was presented that was very similar to that used in the past by some members of WGMS. It was based on the regression of contaminant data and 'normalizer' data from a set of co-genetic samples. Deviations from the background relation were considered to be the anthropogenic portion. However, it was pointed out that with very sandy samples, the calculation of this anthropogenic portion suffers from errors in the measurement of both contaminant and normalizer, but also due to the natural variability of the background. At low normalizer values, the anthropogenic portion cannot be estimated significantly.

After discussion, the ACME felt that the results at the present stage of the ongoing international QUASH project are very promising, and may provide outstanding and long-awaited contributions to the discussion on normalization, as well as giving practical experience in several normalization techniques.

*Need for further research or additional data*

As a result of the promising outcome of the QUASH project, the ACME agreed with WGMS that the revision of Technical Annex 2 of the Sediment Guidelines on normalization should take into account the results of the forthcoming QUASH report.

## 5.6 Statistical Aspects of Monitoring

### 5.6.1 Statistical methods for designing and assessing monitoring programmes

*Request*

There is no specific request; this is part of the continuing ICES work to provide advice on the development of effective methods for designing monitoring strategies and assessing temporal monitoring data.

*Source of the information presented*

The 1999 report of the Working Group on Statistical Aspects of Environmental Monitoring (WGSAEM) and ACME deliberations.

*Status/background information*

The ACME noted that in 1999 WGSAEM considered several statistical methods and analyses for designing and assessing monitoring programmes:

1) dynamic sampling strategies;
2) estimating between-year variance components;
3) preliminary analysis of VIC data.

These are described below.

**Dynamic sampling strategies**

Dynamic sampling strategies are appropriate when there are several monitoring sites but insufficient resources to sample at each site every year. One way of dealing with this is to reduce the frequency of monitoring at each site to, e.g., every other year. An alternative is to dynamically allocate samples to sites where the perceived need for monitoring is greatest. For example, it might be more important to monitor at a site where levels are close to a reference value, than at a site where levels are well below the reference value.

An example of the application of dynamic sampling strategies was considered. The dynamic strategies used smoothers to predict levels at each site in the next monitoring year. Samples were then allocated to sites so as to minimize the risk of incorrectly inferring that levels at each site were above or below the reference value. The dynamic strategies generally outperformed fixed sampling strategies (with the same total sampling resource). Different types and sizes of smoothing windows were considered. It was generally better to have a window that was too small (i.e., to overfit to the data), than one that was too large. However, the results indicated a problem with the current implementation. Intuitively, one would generally expect to allocate more samples to a site where levels change rapidly about the reference level than to a site where levels change slowly. Unfortunately, the opposite was found. The most likely explanation is that the sampling allocation does not take account of the uncertainties in the one-year-ahead predictions.

**Estimating between-year variance components**

If there are only a few years of data available, the question arises as to whether the number of years of data allows 'adequate' estimates of the between-year variance in an annual monitoring programme. Some simulations were run to investigate this issue.

Suppose that $n$ samples are taken each year, in years $t = 1 \ldots T$. Let $y_{tj}$ be the measurement in sample $j$ in year $t$. These measurements might be contaminant concentrations, biomarker measurements, or any other measurements that satisfy the distributional assumptions below. Assume that

$$y_{tj} = \mu + \delta_t + \varepsilon_{tj}$$

where

$\mu$ is the mean level over time,

$\delta_t$ are independent Normally distributed errors with zero mean and variance $\sigma_b^2$,

$\varepsilon_{tj}$ are independent Normally distributed errors with zero mean and variance $\sigma_w^2$.

Here, $\sigma_w^2$ and $\sigma_b^2$ are the within-year and between-year variance components, respectively. Let $\hat{\sigma}_w^2$ and $\hat{\sigma}_b^2$ denote the corresponding residual maximum likelihood estimators.

Table 5.6.1.1 gives the mean, % coefficient of variation, and 5 % and 95 % quantiles of $\hat{\sigma}_b^2$ for different values of $n$, $T$, and $\sigma_b^2$, with $\sigma_w^2$ fixed equal to unity. Each set of summary statistics is based on 100 simulations, so the table gives only a rough indication of the distribution of $\hat{\sigma}_b^2$.

The table indicates the following:

1) Even with ten years of data, $\hat{\sigma}_b^2$ is alarmingly imprecise.

2) Increasing the sample size each year from $n$ = 10 to $n$ = 20 has only a small impact on the distribution of $\hat{\sigma}_b^2$.

3) When $\sigma_b^2$ is 'small' relative to $\sigma_w^2$, negative estimates of $\sigma_b^2$ are quite common, even with ten years of data.

4) When $\sigma_b^2$ is of the same order of magnitude as $\sigma_w^2$, negative estimates of $\sigma_b^2$ are unlikely.

The implication of these results is that we must be very careful about over-interpreting between-year variance estimates, even if there are ten years of data. In particular, a single ten-year time series is likely to be *completely inadequate for making generic* recommendations about the power of a contaminant or biomarker monitoring programme. So what do we do in practice if long time series are unavailable? The answer must lie in combining information from several (many) shorter time series. For example, Nicholson *et al.* (1997) combined information from several time series to explore the likely range of the between-year variance in mercury levels in biota.

**Preliminary analysis of VIC data**

The OSPAR Commission Ad Hoc Working Group on Monitoring (MON) meeting in 1996 proposed a simple international programme called Voluntary International Contaminant (VIC) monitoring for temporal trends. It involves supplemental analyses to the OSPAR Joint Assessment and Monitoring Programme (JAMP) component on trends in contaminant concentrations in biota to obtain quantitative information on the variability in time and space within the guidelines of the sampling strategy. Over three to four years, the participating countries in the VIC programme (Germany, the Netherlands, Norway, and Sweden) have conducted multiple sampling each year at selected stations, at more than one location and/or at different times within the sampling season. The aim was to test sampling strategies for a cooperative revision of the guidelines by 1999.

In a preliminary analysis of data on selected contaminants in cod under the Norwegian VIC, the contaminant levels were adjusted for biological within-sample variation by using a set of largely uncorrelated physiological covariates. The covariates are derived from the original measures by a controlled procedure. This has the advantage of giving covariates that are simpler and easier to interpret than covariates produced by Principal Component Analysis (PCA) or canonical correlation. Covariate regression based on all available Norwegian cod data explains from 20 % to 65 % of the total variance within year and station. The within-sample covariate adjustment for most of the contaminants also reduces the between-sample variance, indicating that the covariate adjustment leads to more comparable values and does not introduce variation.

The preliminary results indicate considerable short-term and local geographical variation for some contaminants, pointing to a possible increase in the power of a temporal monitoring programme for a given monitoring effort by conducting replicate sampling within a year and station defined on an appropriate spatial scale. However, final conclusions have to await a complete analysis based on all data, and taking cost considerations and error distributions of estimated variance components into account.

*Need for further research or additional data*

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

**Reference**

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

**Table 5.6.1.** Values of the mean, % coefficient of variation, and 5 % and 95 % quantiles of $\hat{\sigma}_b^2$ for different values of $n$, $T$, and $\sigma_b^2$, with $\sigma_w^2$ fixed equal to unity.

| | | $n$ = 10 | | | | $n$ = 20 | | | |
|---|---|---|---|---|---|---|---|---|---|
| $\sigma_b^2$ | T | 5 % | mean | 95 % | CV | 5 % | Mean | 95 % | CV |
| 0.1 | 4 | –0.07 | 0.13 | 0.60 | 220 | –0.03 | 0.11 | 0.34 | 150 |
| | 7 | –0.05 | 0.14 | 0.43 | 160 | 0.00 | 0.10 | 0.27 | 80 |
| | 10 | –0.03 | 0.11 | 0.30 | 110 | 0.01 | 0.09 | 0.20 | 60 |
| 1.0 | 4 | 0.08 | 1.11 | 2.54 | 79 | 0.06 | 1.07 | 3.65 | 103 |
| | 7 | 0.10 | 1.05 | 2.28 | 68 | 0.26 | 0.97 | 2.03 | 54 |
| | 10 | 0.27 | 0.96 | 1.79 | 49 | 0.37 | 1.03 | 1.80 | 49 |
| 10.0 | 4 | 0.9 | 8.9 | 21.4 | 74 | 1.8 | 9.7 | 26.6 | 75 |
| | 7 | 2.8 | 10.2 | 23.4 | 65 | 3.6 | 9.6 | 21.6 | 56 |
| | 10 | 3.6 | 9.0 | 17.6 | 44 | 3.4 | 9.7 | 18.0 | 50 |

### 5.6.2 Determination of the number of replicate samples of sediments or biota to characterize an area

*Request*

There is no specific request at the present time; however, the issue originally arose in a request from the OSPAR Commission concerning the development of guidelines for monitoring PAHs in biota and sediments.

*Source of the information presented*

The 1999 report of the Working Group on Statistical Aspects of Environmental Monitoring (WGSAEM) and ACME deliberations.

*Status/background information*

The ACME requested WGSAEM to give advice at its 1999 meeting on the number of replicate samples of sediments or biota needed in order to characterize an area. General principles relevant to this issue are given in ICES (1995) for several different objectives, including estimating an area mean, mapping of contaminant concentrations, identification of hot spots, and detection of temporal changes. However, a specific response on this topic would depend on the underlying objectives, sampling scheme, choice of measurement, and, ultimately, estimates of appropriate components of variability of the monitoring programme. The ACME reviewed and accepted the WGSAEM treatment of this topic presented below.

The usual inference problem in sampling is to estimate some summary characteristic of the population, e.g., the mean density of a macrobenthic species, after observing only the sample. Sampling consists of selecting some part of a population (e.g., a macrobenthic species or an organic contaminant in the sediment in some area) to observe so that one may estimate something about the entire population. With the term 'population' is meant the statistical population. Additionally, one would like to be able to assess the accuracy or confidence associated with the estimates. The uncertainty is a direct result of the fact that only part of the population is observed. When designing a sampling programme, it is necessary to define the units that make up the population of interest. These units can be defined in many ways depending on the study objectives, the type of measurement to be made, costs, etc. The way of selecting those units in the area of interest, the population, is determined by the sampling design. For example, with simple random sampling, with sample size $n$, all units have the same probability of being included in the sample. The random selection of the units of the sample guarantees that the estimate of the population mean is unbiased. The estimate is unbiased because its expected value over all possible samples that might be selected with the design equals the actual population value. Also, the estimate of the variability is unbiased, which can be used to assess the reliability of the survey. By using a properly defined sampling design and suitable estimation methods, several goals such as unbiased estimates, low-variance estimates, and cost efficiency can be satisfied.

**Example: North Sea Benthos Survey**

An example of how to design a sampling scheme is presented here based on the North Sea Benthos Survey. The objective of the survey was to estimate the mean density of several macrobenthic species in the Dutch part of the North Sea with a coefficient of variation for the mean ($CV_{\bar{x}}$) of 10 %. Three species have been chosen for this example. Within this area ($10^6$ $m^2$), several 'sites' (100 $m^2$ each) were 'randomly' selected and at each site several box-core samples (0.068 $m^2$) were also taken randomly. Within this framework, inferences about the variability between and within sites could be made. The cost for visiting a site was about three times higher than the cost of sorting out the samples.

The sampling design

Simple random sampling was chosen because at the start of the research no information about the distribution of the macrobenthic species was available. A two-stage sampling design was used in which first a sample of primary units (the sites) is selected and, secondly, a 'random' sample of secondary units (the box-core samples) is taken from each of the selected primary units. Let $N$ denote the number of primary units in the population and $M_i$ the number of the secondary units in the $i$th primary unit. For simplicity, let us assume that

$$M_1 = M_2 = \cdots = M_N = M .$$

As a pilot study, 25 sites ($n = 25$) were chosen and 5 samples at each site ($m = 5$) were taken. Let $x_{ij}$ denote the abundance of a macrobenthic species for the $j$th sample at the $i$th site. The population total is $\tau = \sum_{i=1}^{N}\sum_{j=1}^{M} x_{ij}$ . The total number of the potential units available for measurement from the population is $NM$. The true mean for these $NM$ units is $\mu = \tau / NM$ . The true mean per secondary unit in the $i$th primary unit is

$$\mu_i = 1/M \sum_{j=1}^{M} x_{ij} = x_{i.}/M .$$

Then the unbiased estimates for $\mu$ and $\mu_i$ are the arithmetic mean of the $mn$ measurements $x_{ij}$ of the sample:

$$\bar{x} = \frac{1}{nm}\sum_{i=1}^{n}\sum_{j=1}^{m} x_{ij} = \frac{1}{n}\sum_{i=1}^{n} \bar{x}_i . \qquad (1)$$

Then the uncertainty in $x$ due to only $nm$ of $NM$ secondary units being measured (the sampling variance) is estimated (Cochran, 1977, p. 276) by:

$$\sigma_{\bar{x}}^2=(1-f_N)\frac{\sigma_b^2}{n}+(1-f_M)\frac{\sigma_w^2}{nm}, \quad (2)$$

where $f_N = n/N$, $f_M = m/M$; $(1-f_N)$ and $(1-f_M)$ are called the finite population corrections, and

$$\sigma_b^2=\frac{1}{N-1}\sum_{i=1}^{N}(\bar{x}_i-\bar{x})^2, \quad (3)$$

the between-station variance, and

$$\sigma_w^2=\frac{1}{N(M-1)}\sum_{i=1}^{N}\sum_{j=1}^{M}(x_{ij}-\bar{x}_i)^2, \quad (4)$$

the within-station variance.

In this example, the finite population corrections have been neglected.

The usual approach to analyse these data and to obtain estimates for the above variances is to use analysis of variance (ANOVA). In this case, a nested analysis of variance was used because the samples (secondary units) are nested within sites (the primary units). Before we can proceed with the computation, we should formulate a model of the nested ANOVA. The equation is

$$y_{ij} = \mu + \alpha_i + \varepsilon_{ij}, \quad (5)$$

where

$y_{ij}$ is the ln-transformed density,

$\mu$ is the grand mean,

$\alpha_i$ is Normally, independently, and identically distributed according to $N(0,\sigma_b^2)$,

$\varepsilon_{ij}$ is also Normally, independently, and identically distributed according to $N(0,\sigma_w^2)$ and $\alpha_i$ and $\varepsilon_{ij}$ are independent.

The densities of the macrobenthic species are ln-transformed according to ln(density+1/0.068) to satisfy the assumption of a Normal distribution with constant variance between sites and within the site. The addition of 1/0.068 is used for the zero densities and to account for the area of the box-core sample.

Model (5) has two variance component parameters to be estimated. The ANOVA table (Table 5.6.2.1) gives insight into the sums of squares and the expected mean squares.

The estimators for the variances $\sigma_b^2$ and $\sigma_w^2$ are therefore

$$s_b^2 = (MSS - MSW)/m,$$

$$s_w^2 = MSW.$$

We are using $s_b^2$ and $s_w^2$ to give the difference between the estimators and the population parameters.

From this pilot study, we have estimates of the mean and the variances and we could ask, e.g., how to choose $m$ sites and $n$ replicates to achieve a given coefficient of variation of the mean and what is the least expensive design to do so.

To answer this question, the following steps have to be taken:

Estimate the expected variance of the area mean density of a species ($\bar{y}$) based on $n$ sites and $m$ subsamples:

$$s_{\bar{y}}^2 = \frac{s_b^2}{n}+\frac{s_w^2}{nm}. \quad (6)$$

**Table 5.6.2.1.** The analysis of variance table for a one-way nested ANOVA.

| Source | Sum of squares | Df | Mean squares (MS) | Expected MS |
|---|---|---|---|---|
| Sites | $\sum_i^n(\bar{y}_i-\bar{y})^2$ | $n-1$ | MSS | $\sigma_w^2+m\sigma_b^2$ |
| Within-site | $\sum_i^n\sum_j^m(y_{ij}-\bar{y}_i)^2$ | $n(m-1)$ | MSW | $\sigma_w^2$ |
| Total | $\sum_i^n\sum_j^m(y_{ij}-\bar{y})^2$ | $nm-1$ | MST | |

For a clarification of this formula, we have to go back to formula (5). The overall mean is namely based on the mean of $n$ drawings of $\alpha_i$ and $nm$ drawings of $\varepsilon_{ij}$, with respective variances $\sigma_b^2/n$ and $\sigma_w^2/nm$.

The choices between $n$ and $m$ are made iteratively. We could rephrase formula (6) as:

$$n=\frac{1}{s_{\bar{y}}^2}\left(s_b^2+\frac{s_w^2}{m}\right). \qquad (7)$$

The $100\sqrt{s_{\bar{y}}^2}$ is approximately the % coefficient of variation of the mean density on the original scale ($\bar{x}$). The original mean, assuming log-Normality, could be calculated by

$$\bar{x}=\exp(\bar{y}+s_{\bar{y}}^2/2). \qquad (8)$$

Actually, by back transforming $\bar{y}$ with $\exp(\bar{y})$, the median of the original values is calculated.

The estimates of the mean, the between-site variance and the within-site variance of the density for the three species are shown in Table 5.6.2.2. It is clear that for *Amphiura filiformis* and *Lanice conchilega* the main variation is between the sites and not within a site. The coefficient of variation of the mean on the original scale varies between 11 % and 35 %.

**Table 5.6.2.2.** The estimates of the mean $\bar{y}$, the back-transformed mean density $\bar{X}$ (formula 8) in number/m$^2$, the standard error of $\bar{X}$, the between-site variance $s_b^2$ and the within-site variance $s_w^2$ for the ln-transformed densities of three benthic species in the defined area of the North Sea.

| Species | $\bar{y}$ | $\bar{X}$ | $s_{\bar{y}}$ | $s_b^2$ | $s_w^2$ |
|---|---|---|---|---|---|
| *Amphiura filiformis* | 3.45 | 33 | 0.305 | 2.302 | 0.142 |
| *Lanice conchilega* | 4.23 | 73 | 0.346 | 2.817 | 0.898 |
| *Scoloplos armiger* | 3.19 | 24 | 0.113 | 0.283 | 0.178 |

Next, define the cost function:

Suppose the average cost of sampling is described by the simple cost function

$$C=c_1n+c_2nm, \qquad (9)$$

where $C$ is the constraint of the total amount of money available for doing the survey, $c_1$ is the cost for visiting a site and $c_2$ is the cost for sorting out a sample. In this case, the cost for visiting a site is three times higher than that for sorting out a sample, so for this design the cost in some currency is:

$$C=3\times25+25\times5=200$$

Figure 5.6.2.1 shows the results for $n$ and $m$ given the choice of the CV=10 % ($\sqrt{s_{\bar{y}}^2}$ =0.1) for *Amphiura filiformis, Lanice conchilega,* and *Scoloplos armiger.* The lines are flattened because the main variability is caused by the variability in density between the sites and, therefore, the gain in reducing the coefficient of variation for this sampling strategy is to use more sites instead of subsampling. The cost function is the constraint for choosing the $n$ and $m$, and for this example we do not want to cross 200.

**Figure 5.6.2.1.** The different choices of $n$ and $m$ for a CV = 10 % for *Amphiura filiformis, Lanice conchilega,* and *Scoloplos armiger* and the cost function with a constraint of 200.

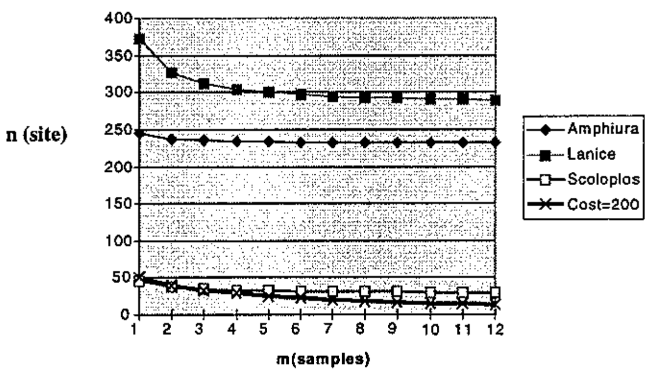



Figure 5.6.2.1 shows that different $n$ and $m$ should be chosen according to the differences in variance for the selected species. To choose a set, a first approach could be to use the maximum or mean of all the $n$'s and $m$'s, because normally we do not know which species is more important. For these three species, the choice based on the maximum would be $n = 371$ and $m = 1$, a design which is not possible because of the lack of money. If we want to continue this programme, a maximum of 50 sites with one sample could be visited. In this case, for *Amphiura filiformis, Lanice conchilega,* and *Scoloplos armiger,* a coefficient of variation for the mean of 22 %, 38 %, and 10 %, respectively, is possible. It would be better to have another sampling design which takes into account the high variability between the sites. A possible approach is stratified random sampling in which the heterogeneous population is divided into sub-populations (strata), each of which is internally homogeneous.

It should be noted that the above objective is a very simple one. Very often the interest is in detecting a trend with a certain power of having a mean or coefficient of variation with a certain confidence.

*Need for further research or additional data*

The ACME agreed that there is a continuing need for the further development of effective statistical methods, supported by an evaluation of their power and sensitivity.

**References**

Cochran, W.G. 1977. Sampling techniques. John Wiley and Sons, New York, NY, USA.

ICES. 1995. Report of the ICES Advisory Committee on the Marine Environment, 1995. ICES Cooperative Research Report, 212: 31–33.


### 5.6.3 Visualization tools for integrating and interpreting biological effects

*Request*

There is no specific request; this is part of the continuing ICES work to provide advice on the development of effective methods for assessing and interpreting monitoring data.

*Source of the information presented*

The 1999 report of the Working Group on Statistical Aspects of Environmental Monitoring (WGSAEM) and ACME deliberations.

*Status/background information*

The ACME took note of the WGSAEM consideration of the use and interpretation of star plots as a visual tool to describe and summarize biological effects data. Multi-biomarker approaches have currently been adopted in an attempt to characterize an environmental impact based on a variety of biological responses. Due to data scarcity, it is sometimes difficult and even impossible to use multivariate analysis techniques or regression models. Even if the data set is sufficiently large, a graphical description can always be considered as a first essential step in statistical analysis. An appropriate linear transformation allows the reduction of biomarker values to the same scale, obtaining scores, and thus permits between-station, between-survey, and even between-biomarker comparisons (star radius = score by station). Results at a given station can then be integrated into a single index, the integrated biomarker response (IBR), which is the area of a star plot where radia correspond to biomarker values measured at that particular station. Eventually, a survey can simply be summarized into a star plot where radius values are IBRs estimated at each station during that particular survey.

These graphical tools were applied to 'early warning signal' bioeffects (glutathion-*S*-transferase (GST), catalase, and acetylcholinesterase (AChE)) measured in blue mussels (*Mytilus edulis*) at six stations (three sites and two stations per site) in the Baltic Sea. In an attempt to obtain an impression of exposure-bioeffect relationships, 'survey' star plots were visually compared to star plots of PAH concentrations measured in tissues from the same individuals. Once data had been transformed, star plots appeared to be a very practical tool to visualize bioeffects and also various interactions existing in the available data set. There is no inference in the proposed plots, but nevertheless, a sense of uncertainty could be represented by plotting on the same plot an inner star joining lower limits of confidence intervals and the corresponding outer star; the risk here could be to lose some clarity in the visual impact.

Star plots have the potential to be misinterpreted by managers and others who are not aware of how they have been derived. For example, non-specialists may be tempted to assume that the area of the plot is proportional to the degree of 'damage'. Such problems could be partially overcome by scaling the area of the various axes in accordance with their perceived importance. However, the general point remains that these representations of data should be primarily used by specialists as a convenient tool for summarizing multifactorial monitoring information.

The ACME noted that star plots can also be used for visualizing the ratio of the current level to some reference or background level.

**Graphical tools exploration**

If star plots are to be used for the illustration of biomarker results, the data should be transformed in a way that allows 'unbiased' visual comparison (1) between sampling stations—the radius represents the score of a given biomarker at the given station, or (2) between biomarkers—the radius represents the score of the given biomarker at a given station.

For a given biomarker, the successive steps of data transformation (referred to as method (0)) are:

a) Compute the mean estimate, say $X$, when individual results are available, otherwise, use the value of the pooled sample.

b) Compute the $X$s mean ($\mu$) and standard deviation ($\sigma$) over all stations and/or surveys, depending on which comparisons are to be made.

c) Normalize $X$, i.e., get $Y$, where $Y=\frac{X-\mu}{\sigma}$.

d) In cases where the bioeffect corresponds to an inhibition, compute $Z = -Y$, versus $Z = Y$ in the opposite case of an activation.

e) Find the minimum value (*Min*) obtained over all $Y$ values, and add it to the $Z$s; finally, obtain the score

$$S = Z + Min, \text{ with } S \geq 0.$$

WGSAEM also investigated methods of data standardization; for biomarker $k$ and station $j$, they can be defined as follows:

Method (1)

$$p_{ijk} = \frac{\overline{X}_{ijk}}{\overline{X}_k^{MAX}},$$

where $\overline{X}_{ijk}$ is biomarker $k$ estimated mean at station $i$ for survey $j$, and $\overline{X}_k^{MAX}$ is biomarker $k$ maximum mean value across all stations and surveys; then the percentage $p_{ijk}$ expresses the effect corresponding to biomarker $k$ relative to the most impacted element among all survey*station combinations. In cases where the bioeffect corresponds to an inhibition instead of an activation, the percentage becomes $p_{ijk} = \frac{\overline{X}_k^{min}}{X_k^{MAX}}$, where $\overline{X}_k^{min}$ is biomarker $k$ minimum mean value across all stations and surveys.

Method (2)

$$q_{ijk} = \frac{\overline{X}_{ijk}}{X_k^{MAX}} \frac{s_k^{MAX}}{s_{ijk}},$$

where $s_{ijk}$ is biomarker $k$ estimated standard deviation at station $i$ for survey $j$, and $s_k^{MAX}$ is biomarker $k$ maximum standard deviation value across all stations and surveys; this weighting allows the between-individual variability to be taken into account. The sample showing the highest variability between mussels (in our case study) for a given biomarker will not have its value changed compared to (1), whereas a very homogeneous batch (but not all animals with the same value) will have its value inflated, and the degree of inflation will be the ratio

$$\frac{s_k^{MAX}}{s_{ijk}}.$$

Method (3)

$$r_{ijk} = \frac{\widetilde{X}_{ijk}}{\widetilde{X}_k^{MAX}},$$

where $\widetilde{X}_{ijk}$ is biomarker $k$ estimated median at station $i$ for survey $j$, and $\widetilde{X}_k^{MAX}$ is biomarker $k$ maximum median value across all stations and surveys.

Specific comments on the investigated methods are:

In method (1) the mean is sensitive to outlying observations, as is the maximum value; however, all values are divided by the same maximum value for a given biomarker. On the other hand, it changes the relative contribution of this biomarker among the other biomarkers.

Method (2), compared to (1), allows between-individual variability to be taken into account and basically penalizes stations with high fluctuations in individual measurements. Whether it is desirable to take this into account depends on other considerations than statistical concepts; ecotoxicologists might not want to discard or even smooth what seem to be outlying observations, but which reflect an actual impact on the marine fauna. The correction factor distorts dramatically the star plot visual aspect (not shown) and should not be considered.

In method (3) the median provides a robust way of estimating the bioeffect value at each station. Again, outlying observations are not considered in this case, which might be an argument not to choose this method.

The use of methods (0), (1), and (3) to summarize survey results is shown in Figure 5.6.3.1 for three surveys at the six sampling stations.

Using the mean (1) or the median (3) appears to be slightly less visually discriminant than using method (0). The main reason for this is that calculating percentages, instead of using a normalization procedure, incorporates the biomarker unknown natural or background level, which can also be variable according to the sampling period within the year. Method (0) seems to better reflect the very high survey*station interactions revealed by an ANOVA test (results not shown). Potential users of such tools should be aware that the impact of the transformations performed on the data are data-set sensitive, but to a lesser extent for method (0) compared to methods (1) or (3). This method could be recommended as a visual tool, potentially useful for scientists in advisory processes related to decision making.

The use of 'half-star' plots was also investigated. Half-star plots give a visual sense of direction, for example, clockwise. To be in accordance with this visual constraint, biomarkers can be ordered according to a short- to long-term bioeffect gradient, i.e., going from early warning signals to biological endpoints (see Figure 5.6.3.2). In the same spirit, early warning signals could be represented this way, as long as the biomarker order on the half circle is meaningful. Half circle sectors should be of the same size to avoid a visual, undesired hierarchy between the various biomarkers or classes of biomarkers. For example, groups of biomarkers reflecting the same stage may be given a sector of the same size as the sector corresponding to a single biological endpoint.

**Figure 5.6.3.1.** Survey summary results for three surveys using methods (0), (1), and (3). K0, K2, W2, W3, P2, and P4 are the six sampling stations.

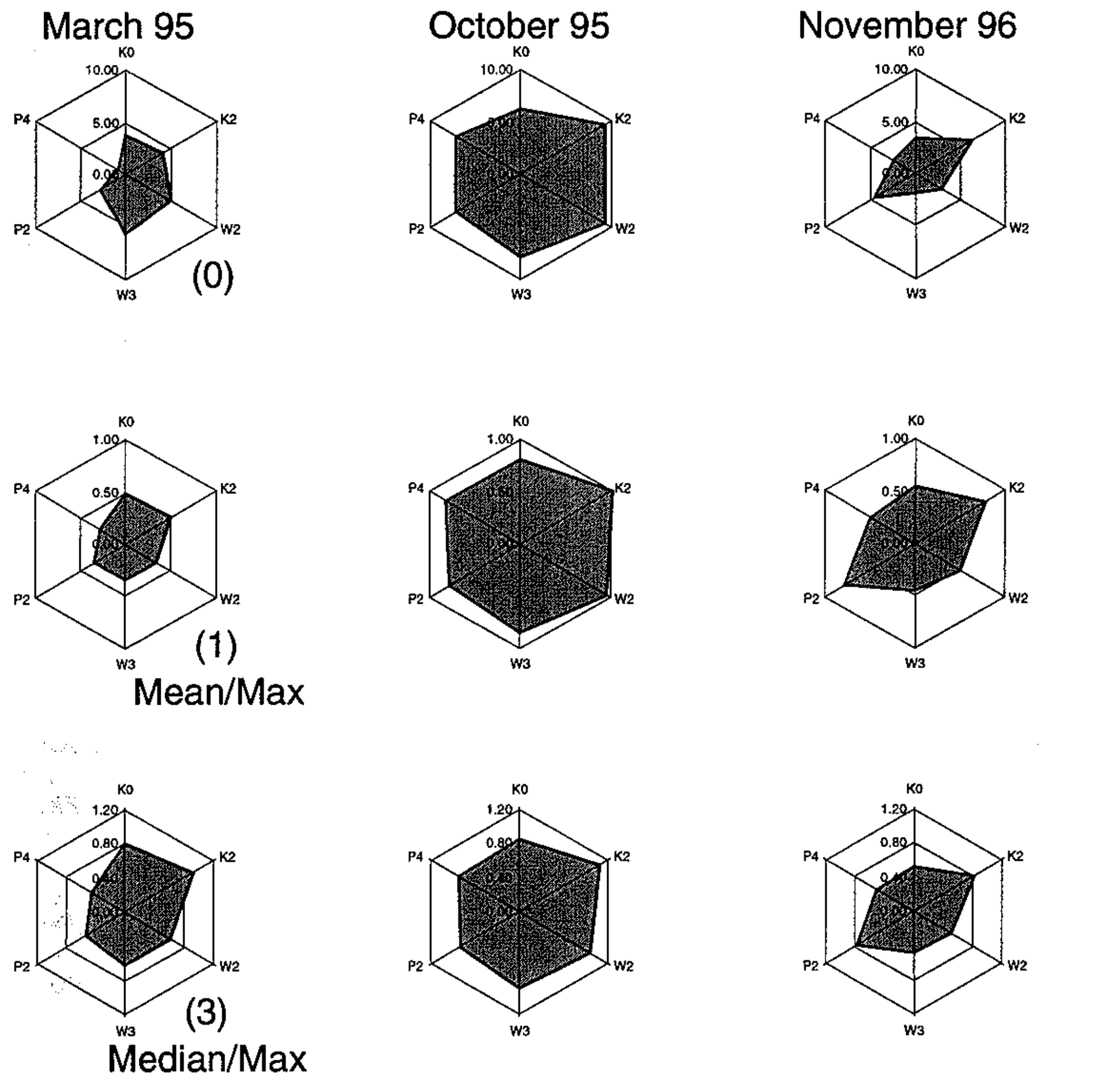
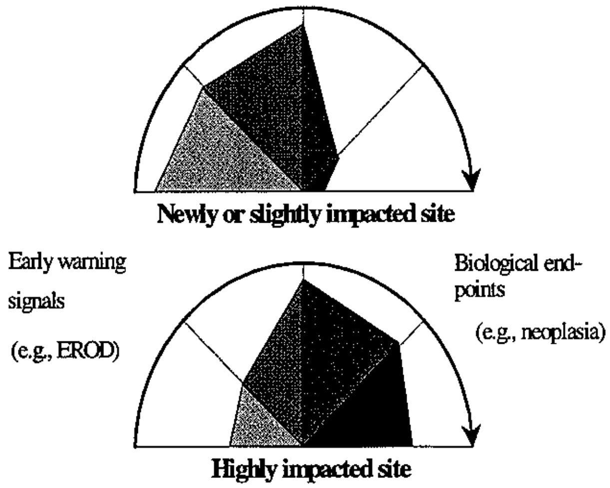



**Figure 5.6.3.2.** Half-star plots used as an aid for site typology representation.




*Need for further research or additional data*

There is a continuing need for the further development of effective visualization tools for integrating and interpreting biological effects. Specifically, it is desirable to continue the development of graphical methods for multivariate data.

### 5.6.4 Implications of the MON98 assessment on the monitoring of temporal trends in contaminants in biota

*Request*

There is no specific request; this is part of the continuing ICES work to provide advice on effective methods for assessing temporal trend monitoring data.

*Source of the information presented*

The 1999 report of the Working Group on Statistical Aspects of Environmental Monitoring (WGSAEM) and ACME deliberations.

*Status/background information*

The statistical method used to assess temporal trends in contaminant concentrations in biota has been developed over several meetings of the OSPAR Ad Hoc Working Group on Monitoring (MON) in cooperation with WGSAEM and based on ICES advice. However, following the general objectives outlined in Section 6.4, below, the ACME suggests further improvements through a 'package of assessment tools'.

The ACME noted that, at the MON98 meeting in February 1998, almost 1300 time series of contaminant data in biota samples from the OSPAR Convention Area were assessed for temporal trends within four regions. New for this assessment were comparisons with BRC/EAC (Background Reference Concentrations/ Ecotoxicological Assessment Criteria) as an assessment tool. The ratio of the estimated average concentration of a contaminant for recent years and the BRC/EAC for that contaminant, where they were available, were computed. To take account of the uncertainty of this estimate, the upper 95 % confidence interval for this ratio was also computed. Minimum detectable trends for a ten-year period at a fixed power of 90 % were also reported for each contaminant.

For each data set with data for seven or more years, the method summarizes trends using a smoother, a non-parametric curve fitted to the median log-concentrations. This summary is supported by a formal statistical test of the significance of the fitted smoother, and by tests of the linear and non-linear components of the trend.

Few statistical assumptions are required for the fitted smoother to be valid. Mainly, the annual contaminant indices should be independent with a constant level of variability. For the statistical tests to be valid, there is a further assumption that the residuals from the fitted model should be log-Normally distributed. The theory and methodology are described in detail in Nicholson *et al.* (1998).

Despite the very large number of data series assessed, many series were rejected through the quality assurance (QA) procedures. It is clearly desirable to remove invalid and meaningless data. But QA data may be used to do this solely because they are available and the procedure is easily expeditable. However, there has been little quantitative evaluation of what benefits are achieved. In practice, it may be that data are being removed unnecessarily. Further, there may be other more important failings in the data which the quality control procedures may not detect (e.g., inconsistencies in the sampling strategy). The real question is whether poor analytical quality makes the assessment misleading. This question can be addressed objectively by considering the effect of analytical accuracy on the mean and variance of the annual contaminant indices observed. An outline of such an evaluation is presented in OSPAR (1998).

Before considering any changes in the current practices of filtering data according to QA criteria, some statistical evaluation should be undertaken to establish whether:

1) the current practice achieves any practical benefits and genuinely improves the quality of the trend assessments;
2) the current practice should be modified to give a more sensible balance between the prescribed standards for analytical variation and the observed levels of between-year and sampling variation;
3) screening for non-analytical violations of the monitoring guidelines should be introduced;
4) the practice of removing data sets should be supplemented or replaced by the use of more robust statistical methods.

*Additional comments*

The ACME expressed its appreciation for the work carried out by WGSAEM on further improvements in the statistical tools for contaminant trend assessments.

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OSPAR. 1998. MON98 Draft Assessment Report (MON 98/6/1-E). OSPAR Commission, London.

Nicholson, M.D., Fryer, R.J., and Larsen, J.R. 1998. Contaminants in marine organisms: A robust method for analysing temporal trends. ICES Techniques in Marine Environmental Sciences, No. 20.

## 5.7 Importance of Long Time Series Data for the Interpretation of Monitoring Data and the Preparation of Assessments

*Request*

There is no specific request; this is part of the continuing ICES work on issues related to monitoring the marine environment.

*Source of the information presented*

The 1999 report of the Working Group on Environmental Assessment and Monitoring Strategies (WGEAMS) and ACME deliberations.

*Status/background information*

The ACME noted that there is a widespread problem of maintaining long time series of monitoring due to cutbacks in funding. In view of the increasing demand for environmental assessments at both the national and international level, this is an unfortunate development. Integrated environmental assessments require a broad range of data on the physical, chemical, and biological conditions in marine ecosystems (Bokn and Skjoldal, 1999). There is, therefore, a clear need to continue and to improve long-term monitoring of the marine environment.

At the Second London Oceans Workshop (London, December 1998), representatives from forty national governments made the following recommendations:

- 'More routine, systematic and long-term observations are required, such as planned by the Global Ocean Observing System. Present monitoring is insufficient to address current problems at ocean, regional and local levels. The scientific skills exist to implement a global observing system spanning ocean basins and providing six months' climate forecasts.
- Efforts to standardize data collection and dissemination need to be encouraged and enhanced.
- Close monitoring of persistent pollutants is needed'.

The Workshop on the Ecosystem Approach to the Management and Protection of the North Sea (NCM, 1998), concluded that the present monitoring of the North Sea is often insufficient to reveal human impacts on the ecosystem. There is a need for improved, integrated monitoring through coordination and harmonization of existing national and international monitoring activities, as well as through implementation of new methods and technology. Monitoring provides updated information about the state of components of the ecosystem, as a basis for assessments and management decisions. The Workshop recognized the need for research to provide basic knowledge and insight into the functioning of the North Sea ecosystem. Important features of the ecosystem dynamics are long-term and large-scale variability related to fluctuations or changes in climatic driving forces. Long-term monitoring provides data on such variability which can be used in research to reveal the underlying mechanisms. This will contribute to solving a major challenge in integrated environmental assessments, which is to separate the influence due to human activities from natural variability.

There are several examples of ecological relationships revealed through long-term monitoring which can be used in assessments and forecasts of fish stocks and environmental conditions.

Svendsen *et al.* (1995) examined the influence of climate on recruitment and migration of fish stocks in the North Sea. They showed that two variables, reflecting wind conditions and areal extent of water masses, could explain more than 70 % of the variance in the recruitment of North Sea whiting, herring, cod and saithe, and in the fraction of western mackerel migrating into the North Sea, based on monitoring time series from the late 1960s to 1990 (Figure 5.7.1). For North Sea herring, the size of the spawning stock biomass was also taken into account since this varied greatly over the time series. A relationship between modelled inflow of Atlantic water and the migration of horse mackerel into the North Sea has also been found (Iversen *et al.*, 1998). These relationships demonstrate that interannual climatic variability has a large impact on the state of North Sea fish stocks.

Changes in dominant zooplankton organisms such as *Calanus finmarchicus* and *C. helgolandicus* have been related to climatic variability as reflected by the North Atlantic Oscillation (NAO) index (Planque and Taylor, 1998). Also, long-term changes in the phytoplankton community, as reflected by the 'greenness index' of the Continuous Plankton Recorder (CPR), have been related to the NAO (Reid *et al.*, 1998a). The abundance of the dinoflagellate *Ceratium* spp. and some zooplankton species has been related to changes in the distribution of water masses (Dickson *et al.*, 1992). Correlations and patterns in time series of zooplankton species from the CPR survey, physical oceanographic conditions, and catches and stock sizes of fish have been revealed by analyses of time series from long-term monitoring. These results strongly suggest that there is a marked climatic influence as well as biological interactions among plankton and fishes of the North Sea ecosystem (Reid *et al.*, 1998b).

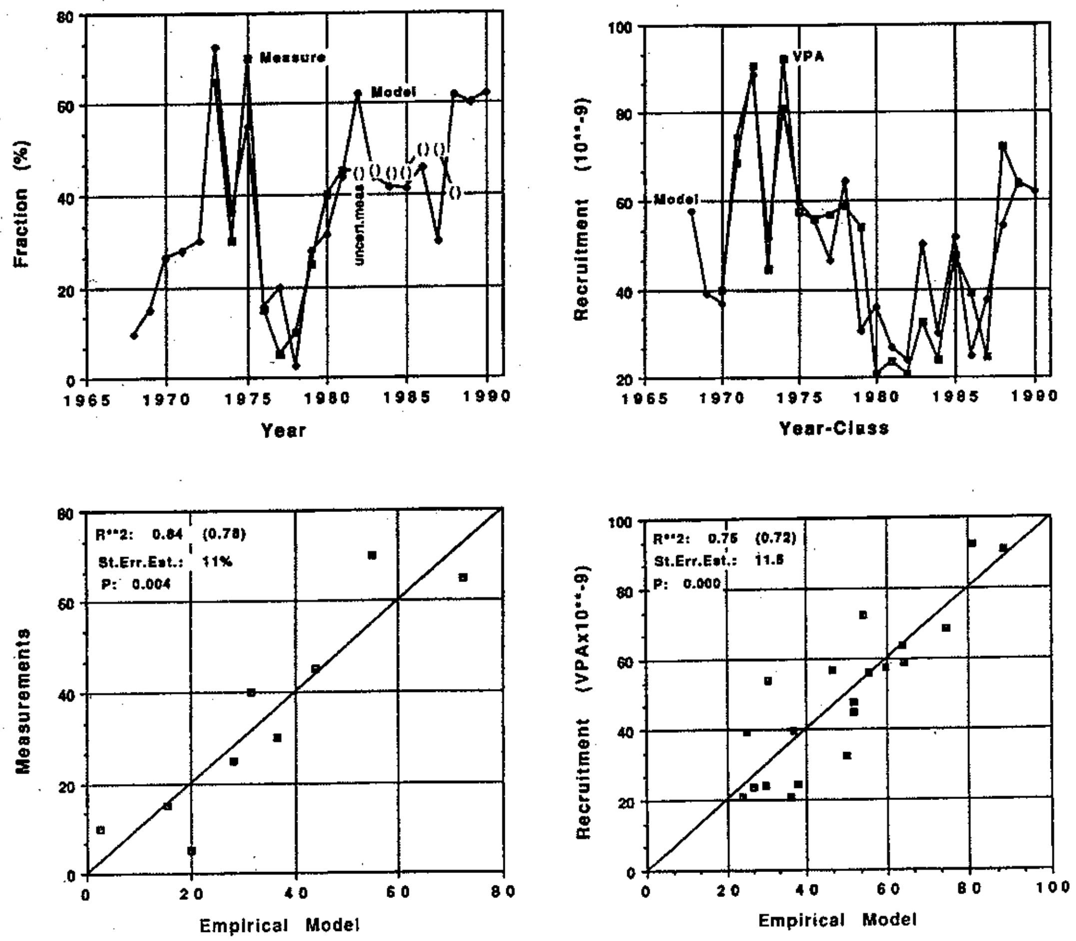
Long-term changes in benthic communities also appear to be influenced by climatic conditions. This has been suggested from the results of monitoring benthic communities in the German Bight, the Kattegat, and off the western coast of Sweden (Kröncke *et al.*, 1998; Tunberg and Nelson, 1998).

*Recommendations*

ICES strongly emphasizes the importance of maintaining long-term monitoring of environmental conditions as a basis for integrated environmental assessments to support environmental and fisheries management systems.

ICES recommends that a review of national and international monitoring programmes on environmental conditions and living marine resources in the ICES area be carried out. This should be done as a basis for considering improvements through coordination and harmonization of existing monitoring activities.

**Figure 5.7.1.** Comparisons as annual times series (upper panels) and scatter plots (lower panels) between measurements and empirical models for the fraction of the western mackerel stocks migrating into the North Sea (left panels) and for the recruitment of 0-group whiting in the North Sea (right panels). The empirical models are based on multiple correlations with two environmental variables, areal extent of Atlantic water in summer and wind effect in the second quarter of the year. From Svendsen *et al.* (1995).

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## 5.8 Strategies for Monitoring Inputs of Nutrients to the Coastal Zone

*Request*

There is no specific request; this is part of the continuing ICES work on techniques and strategies for monitoring programmes, including strategies for monitoring inputs of nutrients to the coastal zone.

*Source of the information presented*

The 1999 reports of the Marine Chemistry Working Group (MCWG) and the Working Group on Shelf Seas Oceanography (WGSSO), and ACME deliberations.

*Status/background information*

The question of monitoring inputs of nutrients to the coastal zone is driven by the increasing problems of eutrophication and is linked to the assessment of the efficacy of measures taken to reduce inputs of nitrogen and phosphorus from land-based sources. It is also of interest for the Commissions dealing with inputs to the convention areas to have a common understanding on the subject. The ACME noted the ongoing work to develop harmonized methods for the calculation and reporting of freshwater-borne nutrient inputs to the marine area in the Harmonized Reporting Procedures (HARP) project within the framework of the OSPAR Commission.

Also of relevance is the input from other sources than riverine inputs, e.g., from the atmosphere and the open sea, both of which can have a significant impact on certain areas. However, only the strategies for the monitoring of riverine inputs to the coastal zone are considered here.

The problem of riverine inputs can also be expressed in terms of gross and net fluxes. The gross flux is the sum of all inputs at the downstream limit of a river; the net flux is the amount that actually reaches the coastal zone. Several questions can be raised, such as: is there a clear definition of the coastal zone? Can the net flux be easily determined? Is there a satisfactory evaluation of the gross flux? Which forms of each nutrient have to be measured?

Nutrients (N, P, Si), present in various forms in fresh water, can be converted from one form to another in the intermediate mixing zone between fresh and coastal waters. Many processes can be responsible for these changes, for example, sedimentation-resuspension, adsorption-desorption, assimilation, mineralization, nitrification, and denitrification. Several of these processes can proceed simultaneously or successively (in space and/or time) while some of them are exclusive (e.g., oxygen-dependent processes).

Many examples, from the literature and from studies undertaken by ICES scientists, show that the morphology of the area under consideration (estuaries, lagoons, deltas) and local hydrographic conditions (tides, wind, currents, stratification, upwelling, etc.) are important factors that affect the overall behaviour of nutrients. While macrotidal estuaries often exhibit conservative behaviours for dissolved inorganic nitrogen, river-lagoon systems in small-tide areas are extremely reactive and complex regarding nutrient fluxes. Inputs can be pulsed or severely influenced by seawater inflow, depending on the location of the boundary selected to determine the fluxes.

From the above considerations, it seems that there will be no unique, simple approach for monitoring nutrient fluxes from estuarine environments. As a consequence, it would seem unrealistic to build a general strategy for monitoring nutrient inputs to the coastal waters. The gross inputs from the river are, at least, what should be well determined. It is important that a recommended procedure is used to calculate the input as well as that all data are stored in an easily accessible database.

In those estuaries where the behaviour of a nutrient has been shown to be conservative, monitoring this nutrient in the terrestrial sources will give the required answer. The nutrient in the estuary has simply to be monitored from time to time to confirm the conservative conditions, since frequent sampling at many stations within the estuary will bring no additional information.

Where (or when) a nutrient is not conservative (but the deviation is not severe), net fluxes may be determined using the nutrient-salinity relationships. By extrapolating from the high-salinity part of the curve to salinity = zero, a net concentration for fresh water is obtained. Multiplying this value by the corresponding water flow gives the net flux. However, a satisfactory determination of the flux will probably require many cruises each year, with many sampling stations.

In general, more complex situations will be encountered. In these cases, high spatial and temporal resolution would be necessary to achieve input determination. However, this would require an unrealistic amount of resources for monitoring such areas.

It is therefore suggested that ecological modelling can advantageously replace high-resolution monitoring programmes. However, models should be refined enough to take into account all major processes that may affect nutrient concentrations. Such sophisticated models are being developed in many countries, but mostly still for research programmes. Improvement of their performance and their user friendliness, as well as their dissemination to the monitoring community, should be strongly encouraged. Indeed, these models can be useful tools for predicting potential eutrophication problems and for computing fluxes through any selected boundary. Nutrient inputs via freshwater into estuaries and coastal waters can also be estimated by a combination of the use of hydrodynamic models and measurements of nutrient concentrations. Hydrodynamic models can, with a proper spatial and temporal resolution, provide realistic descriptions of mixing and water mass budgets. Realistic water budgets can be used along with measurements of nutrient concentrations to provide estimates of nutrient budgets in estuaries and coastal waters. Water and nutrient budgets can together indicate the magnitude and scale of assimilative loss processes of nutrients.

Once a model is calibrated using a high-resolution study, validation will only require low-resolution monitoring. However, the quality and quantity of the data sets used for the calibration and validation of the models are crucial. There are some data sets presently available, e.g., on sea surface temperature and sea level, but more compiled and quality-controlled data are needed for the validation of environmental models in general. Ecological models, in particular, lack good data sets for validation. The importance of choosing a validation technique suitable for the aims of the model was also stressed and it was felt that ICES and its Working Groups can play an important role here to set up a framework where the subject can be discussed.

Although the problem of monitoring inputs to the coastal zone should not be underestimated, the overall complexity of the problem leads to the general recommendation that the gross inputs from a river should at least be well determined and measurements in the coastal zone should focus more on the effects of inputs by means of monitoring adequate eutrophication indicators.

*Need for further research or additional data*

The ACME supported the view of MCWG that work on this item is very important and should continue in cooperation with other relevant ICES Working Groups, e.g., WGSSO. It was suggested that future work should concentrate on developments in special study areas, e.g., the German Bight, the Danish estuaries and/or the Norwegian coast, where it is known that work is already taking place.

## 6 DEVELOPMENT OF TREND DETECTION METHODS

*Request*

Item 4.1 of the 1999 Work Programme from the OSPAR Commission, described in detail in Section 6.1, below.

*Source of the information presented*

The 1999 report of the Working Group on Statistical Aspects of Environmental Monitoring (WGSAEM) and ACME deliberations.

*Status/background information*

The ACME took note that, following the 1997 ICES/OSPAR Workshop on the Identification of Statistical Methods for Trend Detection, WGSAEM discussed in 1997 and 1998 several statistical issues concerning the analysis of contaminant inputs data. Subsequently, the OSPAR Commission requested further advice in order to:

- develop a method for including a power function in the trend detection methods for input data;
- develop and assess statistical methods for dealing with data which are more complex than a series of independent, annual unadjusted loads. The methods should address adjustments to annual and monthly data for, *inter alia*, climatic effects;
- develop provisions for the use of monthly data in the trend detection methods.

OSPAR also agreed to a revised specification for advice on trend assessment tools, as described in Section 6.1, below. This consists of three groups of questions to be considered, with priorities for answering them:

a) fine-tuning of the Trend-y-tector;

b) adjustment of loads;

c) the use of monthly data.

Due to time restrictions, WGSAEM addressed only items a) and b), and they are considered in detail in Sections 6.2 and 6.3, below. In Section 6.4, the identification of the desirable components of a package of trend assessment tools is considered, and a more general package of trend assessment tools is proposed. This builds on the structure of the Trend-y-tector (a suite of methods to estimate trends in yearly data for use by OSPAR groups), but identifies more clearly the role of its individual components. The advantages and disadvantages of different methods for each role can then be seen more clearly. The development of the components of the package is based on experience with the current protocol for inputs and with assessments of trends in contaminants in biota. The recommendations at the end of this section contain two implementations of this package recommended for the analysis of input loads.

### 6.1 Specification for Work on Trend Assessment Tools on the 1999 ICES Work Programme

At the December 1998 meeting of the OSPAR Working Group on Inputs to the Marine Environment (INPUT), a more detailed specification of the request to ICES on trend assessment tools (item 4.1 of the 1999 Work Programme from the OSPAR Commission) was prepared. This is quoted in full below.

'1. OSPAR 1998 agreed that the following request for advice on trend assessment tools should be included in the 1999 ICES Work Programme:

To continue the development of trend detection methods for input data in order:

a) to develop a method for including a power function in the trend detection methods for input data;

b) to develop provisions for the use of monthly data in the trend detection methods (taking into account INPUT(1) 98/14/1, § 6.9, e.ii);

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

2. ASMO 1998 agreed that if this request was included by OSPAR in the 1999 ICES Work Programme, then INPUT(2) 1998 could develop further the specifications detailed in this request on the basis that these did not incur any extra costs.

3. In accordance with the agreement made at ASMO 1998, INPUT(2) 1998 agreed to the following revised specification for advice on trend assessment tools:

**A. Fine tuning of the Trend-*Y*-tector**

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

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

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

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

5) Establish the link between overall significance and individual significance levels.

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

**B. Adjustment of loads**

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

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

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

4) Give advice on how to measure the gain of adjustment.

5) Examine whether and when there is a risk of 'over-adjustment'.

6) Consider whether the use of annual adjusted loads may make the trend analysis of monthly loads redundant.

**C. The use of monthly data**

1) Development of provisions for the use of monthly data in these trend detection methods (taking into account that any recommendations should be based on real need and best scientific judgements and should not be driven purely by statistical considerations (INPUT(2) 98/14/1. §6.9 e.ii).

<u>Relative priority of each of the sections of the above request</u>

1. All the tasks are important.

2. However, should available resources not allow completion of the entire request, the following items should be considered as priority and the minimum request: A 4, 5 and 6 and B 1, 5 and 6.

3. If a positive answer is given for §B.6, then §C of the Work Programme above should be given the lowest priority.'

Due to time restrictions, WGSAEM addressed only items 3.A and 3.B, and they are considered in detail below.

The ACME reviewed the material below and accepted it for transmission to OSPAR in response to items 3.A and 3.B of the detailed specification of item 4.1 of the 1999 Work Programme from OSPAR.

## 6.2 Fine Tuning of the Trend-y-tector

This section contains the responses to the six requests listed in Section 6.1, item 3.A. Throughout this section, it is assumed that the trend assessment methods are intended for annual indices, for example, annual loads or annual adjusted loads.

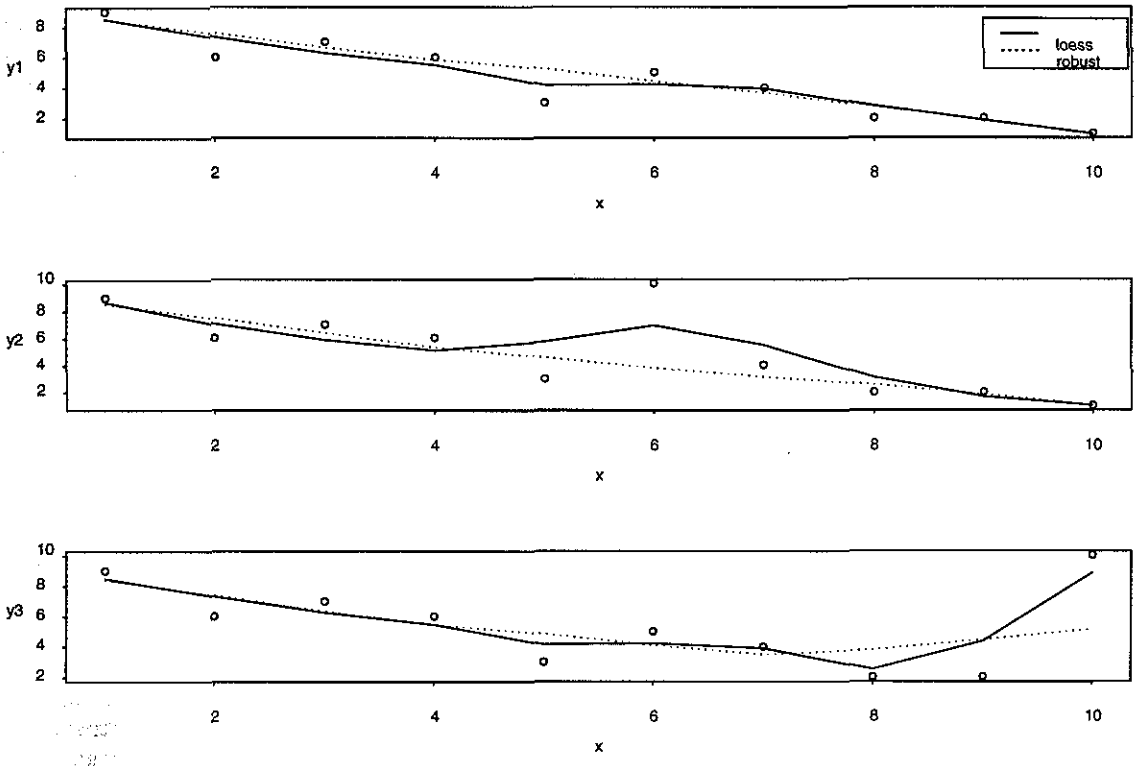
### 6.2.1 Type and specification of the smoother

The current formulation of a smoother-based assessment of trends used for contaminants in biota is a local-linear regression smoother (LOESS) based on a seven-year window. For the number of observations in typical monitoring time series, this gives a sensible number of degrees of freedom for the fitted line, and also appears to have good power over a range of different trend patterns (Fryer and Nicholson, 1999; Annex 1, Part 1). Further, a seven-year window provides a fitted trend which is reasonably smooth, yet can still reveal important short-term changes that may have occurred between assessments (3–5 years).

However, using a fixed window may lead to under- or over-fitting in some cases. Selection of model degrees of freedom by cross-validation is possible, although this does not appear to work well with short data sets (Hastie and Tibshirani, 1990). The alternative is to fine tune the fit for each data set, an option which seems impractical for the analysis of large numbers of data sets. At present, it is difficult to see any alternative to using a fixed window.

A second way to control the smoothness of the fitted curve is to down-weight the influence of individual extreme points. This is achieved by iteratively fitting a weighted smoother, where the weights are determined by the magnitude of individual residuals. Figure 6.2.1.1 shows LOESS and robust smoothers fitted to three data sets. The first is a generally declining series; the second has an isolated high point in the middle of the series; the third has an isolated high point at the end of the series. The robust smoother was constructed using the robust LOWESS smoother in S-plus (equivalent to LOESS) with a window which includes approximately 75 % of the data.

**Figure 6.2.1.1.** The results of fitting LOESS and robust smoothers to three data sets.




As can be seen from the figure, the fitted robust smoother is, of course, more satisfyingly smooth. However, it may not necessarily be a better choice. The robust smoother will be less efficient under assumptions of Normality, and consequently more variable. This could be an important factor which limits its usefulness for short time series. Further, the theoretical basis of the robust smoother is less well developed, and some further development is required.

WGSAEM pointed out that the specification of the smoother will also depend on the underlying error distribution and ultimately on what the smoother will be used for.

### 6.2.2 Calculation of residuals

In the formulation of the smoother-based assessment of trends by Fryer and Nicholson (1999), the residual variance is based on the residuals from the smoother, and hence is directly affected by the fitted smoother—closer fit, smaller variance. Alternative approaches based on differences are less directly related to the degree of smoothing, and aim to provide smaller mean squared error (Dette *et al.*, 1998). Since the estimate of the residual variance has a direct impact on confidence limits of the fitted smoother and in the measurement of power, improved estimates would be desirable. This is an appropriate topic for further development.

Cross-validation may also be used, but whether there would be any advantage is not clear when time series are short. Fryer and Nicholson (1999) also discussed a bootstrap approach to significance tests for the smoother, which may have benefits if the assumption of Normality is not appropriate, or if the assumed F-distribution is not appropriate.

### 6.2.3 Calculation of standard deviation

Steigstra *et al.* (1998, 1999) considered robust methods for estimating the residual standard deviation from a fitted model. In particular, an estimate based on the median absolute deviations was selected for further investigation.

The results of Steigstra *et al.* (1998) showed that care must be taken if the smoother is used with a robust estimate of the residual variance. If a robust smoother is not also used, the smoother will tend to track extreme values, and together with a smaller estimate of the residual variance will provide a very effective test of outliers, giving a significant 'trend' even when no trend is present.

To some extent this problem is addressed in Steigstra *et al.* (1999), where empirical critical values for the contrast between the estimated start and end points of the time series were generated for time series of length 7–30 years and a range of significance levels. These critical values are based on compound error distributions fitted to data for several contaminants in different media, incorporating some outlying values. If future outlying values are of a comparable size, the test will still have the correct significance level when there is no trend.

### 6.2.4 Alternatives to the current smoother test

Within the Trend-y-tector, the implementation of the smoother is restricted to providing a test of a contrast between the estimated means at the start and the end of the time series. Of course, the smoother could also be used to assess the overall trend, with a split between a linear and a non-linear component, as in the OSPAR assessment of trends in contaminants in biota. Selecting an appropriate trend assessment method will be discussed in more detail in Section 6.4, below.

### 6.2.5 Link between the overall and individual significance levels

Within the Trend-y-tector, there is a series of tests (although some are conditional, i.e., only made depending on the outcome of a previous test), each made at a 5 % significance level. However, the significance of the overall test should be controlled at a specified level (e.g., 5 %), with some decision made as to the weight given to each test. A procedure for doing this is demonstrated in Annex 1, Part 2, where two tests, the Mann-Kendall and the non-linear Mann-Kendall tests, are combined to provide an overall test of trend, with each component given equal weight and an overall size of 5 %.

This approach could be applied to the Trend-y-tector, although, as will be discussed in Section 6.4, below, a preferred approach is to separate the components of the Trend-y-tector, and have only a single test of trend, controlled at the desired significance level.

### 6.2.6 Procedure for power calculation, including *post-hoc* power

The procedure for the power calculation is relatively straightforward. ICES (1997), Fryer and Nicholson (1999), and Annex 1, Part 1 provide power curves for different trend patterns and many different statistical tests of trend under the assumption of Normality, including both the Mann-Kendall test and the test based on the seven-year-window smoother. These curves give the power as a function of $k/s$, where $s$ is the residual standard deviation of the yearly index and $k$ is the range between the maximum and minimum mean values.

Hence, the procedure is to obtain an estimate of the residual standard deviation of the yearly index, quantify the magnitude of the effect for an appropriate trend pattern, and then read off the power from the appropriate power curve. The smoother is possibly the best means of providing an estimate of the residual standard deviation, possibly using a modified estimate (see Sections 6.2.1–6.2.3, above).

It should be noted that the published power curves are restricted to time series of 10 and 20 years. They are also restricted to the assumption of Normality, and if one wants to construct power curves for non-Normal distributions, further calculations must be made. Steigstra *et al.* (1998, 1999) constructed power curves using a parametric approximation to empirical distribution. Some indication of the power for intermediate numbers of years could be obtained by linear interpolation, but further power curves could be generated, or more convenient tables provided if the graphs are not adequate. Fryer and Nicholson (1999) give an approximate general expression for the power of linear regression and for the smoother for any number of years and any pattern of trend.

### References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## 6.3 Adjustment of Loads

This section contains responses to the six requests listed in Section 6.1, item 3.B, concerning the adjustment of input loads.

The aim of adjustment is to compensate for the effect of irregular fluctuations induced by varying flow rates, temperatures or precipitation. In order to prevent that climatic influences deteriorate the trend detectability, in the documents OSPAR INPUT(2) 98/5/3 and 98/5/4 it is proposed that an appropriate adjustment of annual and monthly input data be included prior to the application of the trend assessment protocol (see Figure 6.3.1).

**Figure 6.3.1.** Steps of statistical trend analysis of data on riverine and atmospheric inputs.

1 Provision of raw data

2 Adjustment of input load

3 Aggregation of the adjusted data

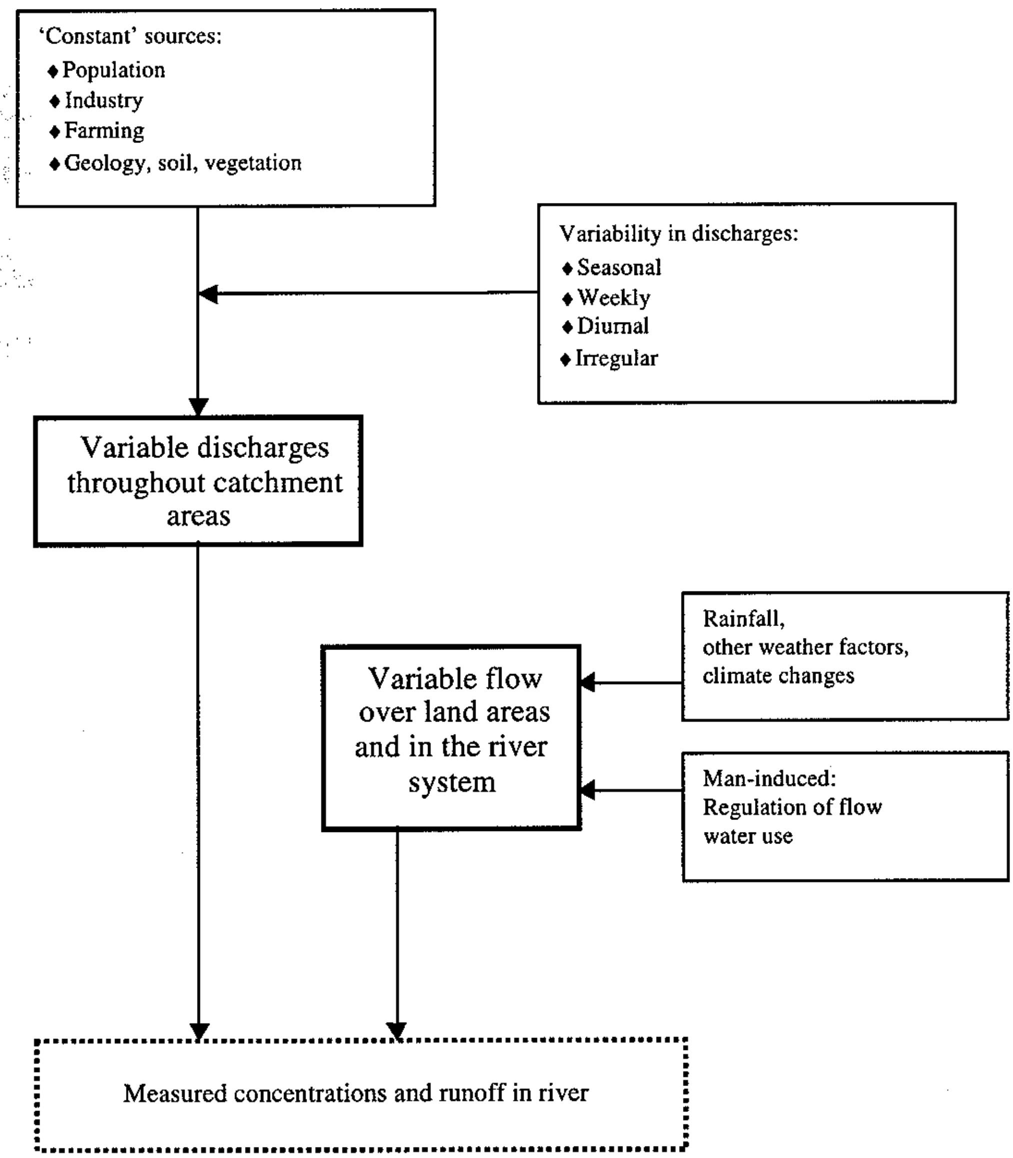
4 Trend detection and trend estimation

5 Power analysis

### 6.3.1 General procedure of adjustment and trend assessment as outlined in documents INPUT(2) 98/5/3 and 98/5/4 with regard to its statistical implications

Considering the statistical implications of adjustment, WGSAEM addressed primarily the question of whether an adjusted input load is only a fictitious quantity or is a quantity with a physical interpretation. In order to obtain a better understanding of adjustment, one may consider Figure 6.3.1.1. The measured input loads are composed of many different contributions, with different variational patterns. They can be seen as caused by a set of more or less fixed sources, varying only slowly. These sources are population, industry, farming, area geology, etc. The long-term changes in discharges (treatment plants, changes in industrial processes, bans/regulations on the use of chemicals) must be seen as part of the description of sources. The discharges from such sources into a catchment area will also vary seasonally, weekly,

**Figure 6.3.1.1.** General relation between sources, discharges, and load in a river system.



diurnally, and by irregular events. The input load will vary both as a result of these variations in discharges, but also because of natural variations in the processes in the river system (varying runoff, retainment and retention, etc.). The river system might be considered as a transport system which transfers nutrients and contaminants from the sources into the sea. The flow partly controls the transport, e.g., in periods of high flow the transport is larger than in periods of low flow. In this system, the flow-adjusted input load may be interpreted as a figure which reflects roughly the emissions/discharges and losses into the transport system.

WGSAEM considered the general procedure of performing adjustment prior to the trend analysis (Figure 6.3.1) as a sensible way of dealing with the issue of adjustment. A main advantage of using this procedure is the flexibility of the approach. An appropriate adjustment procedure reduces the interannual variability of the load values and increases the power of trend tests. A trend estimate based on adjusted loads can be used as a forecast of the load in cases of average runoff.

### 6.3.2 Choice of multiplicative or additive adjustment

Additive adjustment can be exemplified with a linear relationship between the expected riverine load $f_{ij}(q_{ij})$ and the runoff $q_{ij}$ in year $i$ and season $j$:

$$f_j(q) = \alpha + \beta q,$$

where $\alpha$ (intercept) and $\beta$ (slope) are model parameters. With this model structure, flow-adjusted seasonal values $L_{ij,addadjust}$ can be calculated according to:

$$L_{ij,addadjust} = L_{ij} - \left(\left(\alpha + \beta q_{ij}\right) - \left(\alpha + \beta q_{0j}\right)\right) = L_{ij} - \left(q_{ij} - q_{0j}\right)\beta,$$

where $L_{ij}$ denotes the unadjusted load in year $i$ and season $j$ and $q_{0j}$ denotes the long-term average runoff in season $j$. Any model function (not necessarily linear) of the runoff at the same moment or of time-lagged runoff values is, however, possible. Figure 6.3.2.1 demonstrates the relationship between measured and adjusted loads for several runoffs $q$ for additive adjustment in the case of a linear load function. If the actual runoff $q$ is above (below) the long-term average runoff $q_{0j}$, the adjusted load is smaller (larger) than the unadjusted load. Adjusted and unadjusted loads are equal if the actual runoff equals the long-term average runoff.

Apparently, the relative effect of varying flows on the load is very large for small loads (and small concentrations, respectively), whereas this effect is relatively small for large loads and large concentrations. This is not very plausible from a hydrological point of view. Another important drawback of additively adjusted loads is the risk of obtaining negative adjusted loads if both the concentrations and the runoff are small.

These drawbacks can be remedied using multiplicative adjustment, which is based on the following multiplicative model: Let

$$L_{ij} = f_j(q_{ij}) \exp(\varepsilon_{ij}),$$

where

$f_j(q)$ = the mean load in season $j$ at runoff $q$,

$\varepsilon_{ij}$ = random deviation between the logged values $\ln(L_{ij})$ and $\ln(f_j(q))$ in season $j$ and year $i$ caused by non-flow (natural and anthropogenic) influences.

According to this model, the effect of the actual runoff $q_{ij}$ with regard to the load may be expressed as

$$(f_j(q_{ij}) - f_j(q_{0j})) \exp(\varepsilon_{ij}) = L_{ij} - f_j(q_{0j}) \exp(\varepsilon_{ij}).$$

To compensate for this effect, one has to calculate the multiplicatively adjusted load

$$L_{ij,multadjust} = L_{ij} - \left(L_{ij} - f_j\left(q_{0j}\right)\exp\left(\varepsilon_{ij}\right)\right)$$
$$= f_j\left(q_{0j}\right)\exp\left(\varepsilon_{ij}\right) = \frac{f_j\left(q_{0j}\right)}{f_j\left(q_{ij}\right)} L_{ij}.$$

As long as the mean load $f_j$ is specified correctly, negative values for the adjusted load cannot occur. Figure 6.3.2.2 demonstrates the relationship between measured and adjusted loads for several runoffs $q$ for multiplicative adjustment in the case of a linear load function.

Apparently, the relative effect of varying flows on the load is not dependent on the level of the load or the concentration. Moreover, negative adjusted loads cannot occur. Another advantage of multiplicative adjustment is the geometric interpretability of the adjusted load.

This is demonstrated in Figure 6.3.2.3, where the effect of flow adjustments in the case of a linear load function $f(q) = \alpha + \beta q$ is illustrated. The model gives an estimated load $E(L_{ij})$ as a function of flow $q$. The linear load function can be taken as composed of a flow-independent part (i.e., with concentrations decreasing with flow) and a part with loads increasing in proportion to flow (concentrations independent of flow). The load function is shown extrapolated to its intercept with the horizontal axis at $-\gamma = -\alpha/\beta$, with $\beta$ representing the slope of the line. It should be noted that the estimated load at given runoff $q$ can also be written

$$f(q) = \beta q + \beta\gamma.$$

**Figure 6.3.2.1.** Relation between additively adjusted load and unadjusted load.

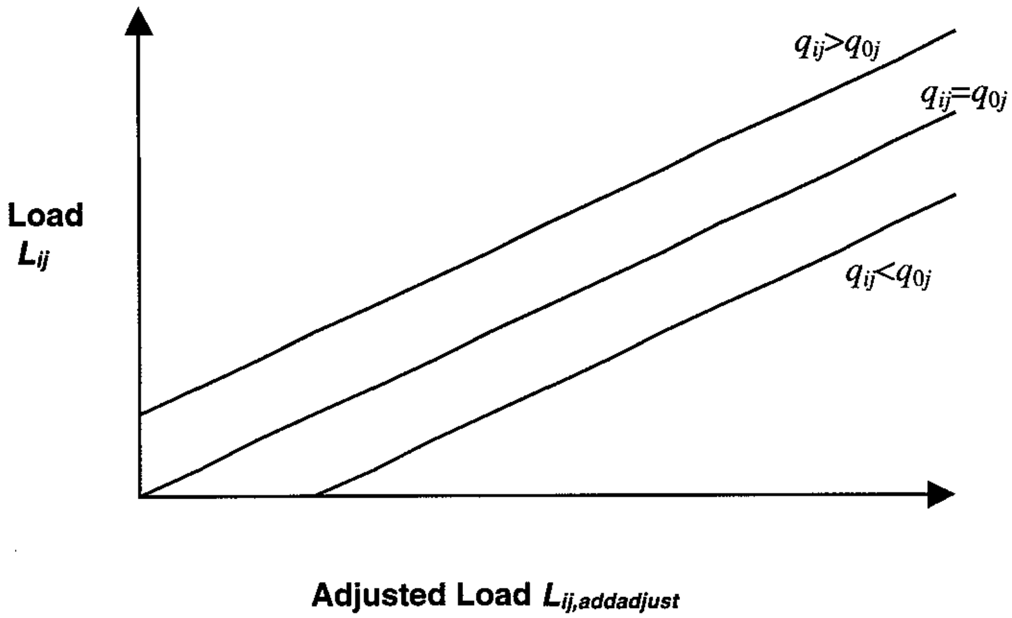



**Figure 6.3.2.2.** Relation between multiplicatively adjusted load and unadjusted load.

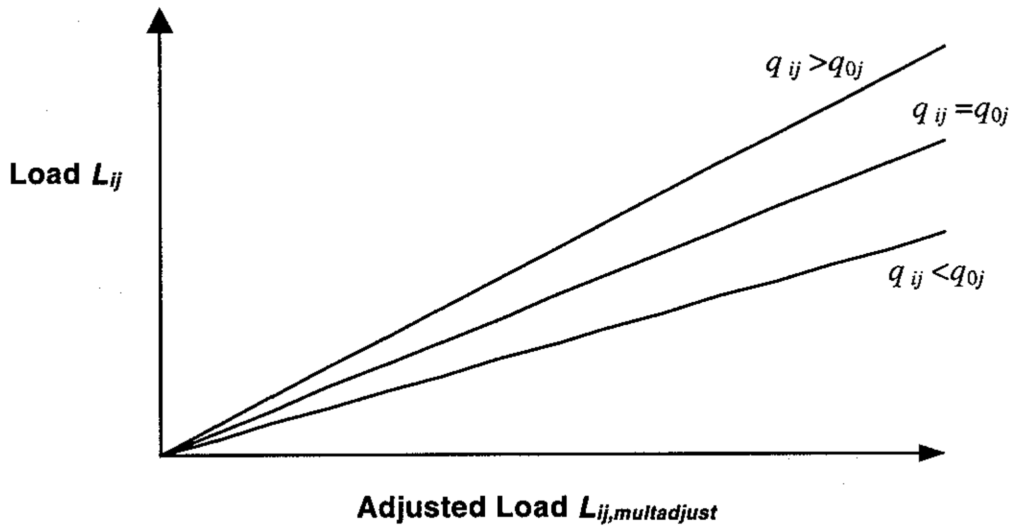
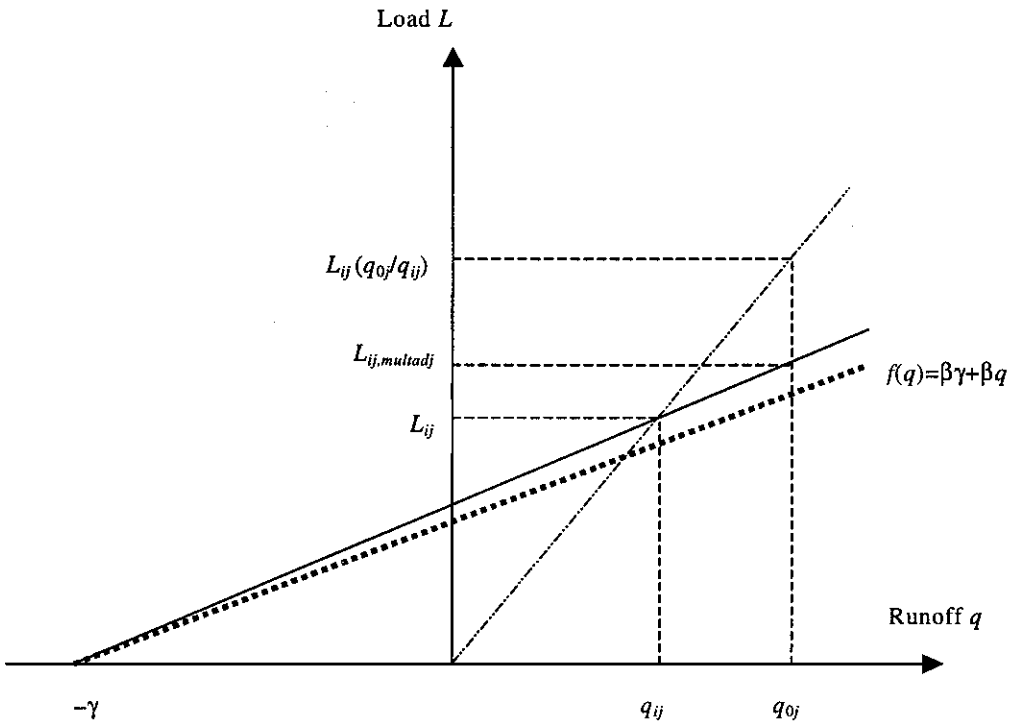


**Figure 6.3.2.3.** Interpretation of multiplicative adjustment in the case of a linear load function.




In the graph, the load function is represented by the dotted line. The observed load $L_{ij}$ at runoff $q_{ij}$ is described by the equation

$$L_{ij} = f(q_{ij})\exp(\varepsilon_{ij}),$$

with variations in discharge appearing as factors of form

$$\exp(\varepsilon_{ij}) = \frac{L_{ij}}{f(q_{ij})}.$$

This variational factor enters as proportional changes of all load values.

In the case of a linear load function, the adjusted load is

$$L_{ij,multadj} = L_{ij}\frac{q_0+\gamma}{q+\gamma}.$$

Under the assumption that the shape of the load function does not change with variations in the discharges, the adjustment of observed loads $L_{ij}$ from observed flows $q$ to mean or normal runoff $q_{0j}$ will give adjusted estimates $L_{ij,adj}$ that better represent the slow changes in discharges over time.

It is seen from the graph that with $\gamma > 0$, the adjusted value will be somewhere between the unadjusted value $L_{ij}$ and the 'naively' proportionally adjusted load $L_{ij}(q_{0j}/q_{ij})$.

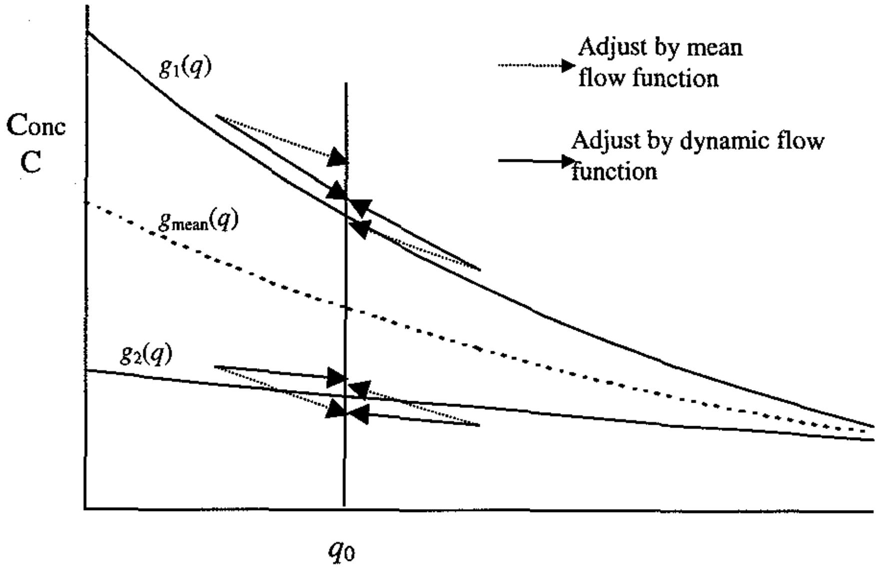
In summary, some drawbacks of additive adjustment are remedied with multiplicative adjustment.

### 6.3.3 Choice of the statistical model and the underlying variables for adjustment

When making adjustments for flow or other factors for trend assessment, one must take into account that different sources will have different response patterns with regard to those factors. Point sources directly into streams and rivers will tend to have a more constant load character, where flow adjustment of loads should not have a large influence. For more diffuse sources, there may be a trend of more constant concentrations, or even increasing concentrations with increasing flows. In all cases, there will normally be considerable variability around the mean relations. Instead of using daily flows, it may be more appropriate to use mean flows and weighted monthly loads as the basis for the statistical method.

For long-term trends that are due to changes in source apportionment (treatment plants, new farming practices), the total response function must be expected to change over time. Therefore, dynamic adjustment with the correct time scale in combination with multiplicative adjustment are expected to function better, whereas non-dynamic adjustment will distort trends. This is demonstrated in Figure 6.3.3.1.

**Figure 6.3.3.1.** Dynamic versus non-dynamic adjustment of concentration flow functions.




The figure shows a hypothetical and very simplified case where two different concentration (c) flow functions $g_1(q)$ and $g_2(q)$ apply to two different source/discharge situations (e.g., before and after abatement measures). The upper function $g_1(q)$ is intended to show a situation with large anthropogenic influence, with a dominating dilution effect of decreasing concentrations with increasing flows, while the lower function may represent a situation with a larger portion of natural discharges of more flow-independent concentrations. Empirical non-dynamic adjustment based on data from a period covering both situations will adjust concentrations along some intermediate function, shown with a dashed line in the graph. For data that really follow $g_1(q)$, this means that adjusted loads for low flows will be overestimated, and for high flows underestimated, while the opposite is true for discharge situations described by $g_2(q)$ (dashed arrows from measured concentration and flow to concentrations adjusted to mean or normal flow). Thus, if there are long-term changes in flow coinciding with the change in discharges, the apparent trends may be distorted. Also, within each of the two discharge situations, trends may be introduced by slow changes in flow.

For both non-dynamic and dynamic adjustment, there is a problem with differences between short-term or long-term variation: This is obvious for retainment/flushing. One possible procedure consists of analysing both unadjusted loads and loads adjusted by various methods. If the results are consistent, the trend must be considered to be robust. If the methods give different results, inferences must be made with caution. In that case, the runoffs or other factors should be analysed separately for trends, to see whether the load trends may be artefacts. It must be pointed out that an appropriate choice of the time window that is used for the calculation of the model parameters is quite important to obtain sensible adjustment results. This especially concerns the years at the beginning and at the end of the time series, since for them only data after and before, respectively, can be taken into account. Therefore, trends appearing at the end of the adjusted time series should be considered with particular caution.

In summary, since apparent trends may be distorted by non-dynamic adjustment, dynamic adjustment might be more appropriate. An appropriate choice of the time window that is used for the calculation of the model parameters is quite important to obtain sensible adjustment results.

### 6.3.4 How to measure the gain of adjustment

A way of measuring the gain of adjustment is to calculate the power of a trend test for a specified trend. Since this would require imposing additional statistical assumptions concerning the stochastic structure of the multivariate time series (e.g., flow, temperature and concentration), it is considered to be more appropriate to substitute the power of detecting a trend by a non-parametric measure of the smoothness of the annual (aggregated) loads. The smoothness of the series can be measured by the standard deviation (or the square root of the mean of the squares) of the second order differences of the logarithmic loads. The outcome can also be interpreted as an estimate of the average rate of relative changes in the trend from year to year.

It is expected that there is a close relationship between the smoothness of the annual loads and the power of detecting a trend in these loads if the smoother test proposed by Fryer and Nicholson is used, i.e., low interannual variability indicates high power for detecting temporal trends. The results of preliminary adjustment analyses suggest that considerable gains in terms of interannual variability and hence in the power of detecting trends can be achieved.

The smoothness of the time series is not the only statistical criterion for the assessment of the adjustment method. The fitted statistical model should also provide a satisfying fit of the concentration data, in order to ensure that model bias is minimized.

In summary, it is recommended to measure the gain of adjustment primarily in terms of the smoothness of the series of annual aggregated adjusted loads.

#### 6.3.5 Examination of whether and when there is a risk of 'over-adjustment'

Over-adjustment may arise if inappropriate statistical models are used. This may be examined by an analysis of the residuals of the statistical analysis. As long as the distribution of the residuals is similar to white noise, no overfitting and no over-adjustment is to be expected.

One should take into consideration that by adjusting loads for long-term trends the influence of long-term changes in runoffs might be removed. This may introduce artefacts in the trends. In order to correct for that effect, it is recommended to calculate adjusted loads not on the basis of the overall mean, but on a running mean of the runoff.

Thus, in order to avoid over-adjustment, it is recommended (1) to check the fitted model by an analysis of the residuals, and (2) to calculate adjusted loads not on the basis of the overall mean, but on a *running mean of the runoff.*

#### 6.3.6 Consideration of whether the use of annual adjusted loads may make the trend analysis of monthly loads redundant

This is still unresolved. Theoretical considerations indicate that due to high between-year correlations of the yearly runoff, unadjusted loads may be highly correlated. An analysis of adjusted loads may therefore increase the power of detecting trends considerably, and the autocovariance of adjusted loads is expected to be reduced. However, preliminary empirical case studies indicate that adjusted loads might also be highly autocorrelated, indicating that an analysis based on monthly adjusted loads might be more powerful (in terms of the power of a trend test) than an analysis on a yearly basis.

More empirical investigations and comprehensive simulation studies are necessary in order to examine the redundancy of analysis of monthly data.

### 6.4 Package of Assessment Tools with Guidelines for Choice and Use

The statistical method used to assess trends must respond to the need to be:

- robust, i.e., to be both routinely applicable to many data sets, and to be as insensitive as possible to statistical assumptions and adverse numerical features such as extreme data values, partial bulking of samples, and less-than values;
- intuitive, i.e., for the results of the analysis to be understandable without a detailed understanding of statistical theory;
- revealing, i.e., to provide easy access to several layers of information about the major features of the data, such as evidence of simple trends, extreme values, missing values, etc.

Following these general objectives and based on lengthy experience with assessments of trends of contaminants in biota and limited experience for inputs, as well as their own expertise, WGSAEM deliberated on what would be the desirable components of a package of trend assessment tools. The consensus was that four separate but complementary components could be identified:

1) graphical presentation of the time series with, for example, a summary line to indicate the general trend and tolerance lines to reveal potential data anomalies;
2) a formal test of trend, with trend defined in an appropriate way for the context of the assessment, and possibly with a power curve which reflects the detectability of the given trend;
3) a measure of the tendency to increase or decrease;
4) a comparison of the current level against some reference level or a level in a previous year.

Some of components 2), 3), and 4), but especially 3) and 4), might then be combined in a statistical meta-analysis to provide summaries across regions, or across contaminants, etc.

These components are already included in the Trend-y-tector, but they are bundled together into a single test of trend. WGSAEM considered that separating the components allowed them to fulfill a more informative role, and also gave greater flexibility in choosing a statistical method for each component that is most appropriate for the purpose of a specific monitoring programme. It also makes it easier for the tools adopted for a specific programme to evolve to meet the particular needs of the assessment group. These needs are likely to be clarified with experience in using the package over a series of assessments.

As before, we assume a series of annual indices, i.e., when the indices are based on underlying raw data, it is assumed that correct processing has taken place, for example, that the indices are adjusted yearly loads, or the logarithms of median concentrations measured in fish.

The following sections discuss each component in the proposed package in detail. Each section summarizes the main purpose of the component; the type of result/information that is produced; and a review of the possible methods, guidelines to when they are appropriate, and a summary of their strengths and weaknesses.

### 6.4.1 Graphical presentation of time series

**Objectives**

A graphical presentation provides a visual summary of the general trend in the data, together with tolerance lines (pointwise prediction intervals) to reveal extreme values, and an opportunity to reveal missing values and other inconsistencies in the data. There may be further advantages of presenting time series grouped by region, by contaminant, or by originating country. This will provide a further opportunity to identify common trends, or common data anomalies, e.g., a consistent extreme value in a given year.

**Type of result or information**

The display should consist of a simple scatterplot of index against year, together with a line capturing the general flow of the data. Forecasts and reference lines referring to quality standards, and pointwise confidence or prediction limits can be added. A linear trend line may also be useful as a reference against which to assess the degree of linearity in the data.

**Guideline to methods, their strengths and weaknesses**

A smoother should be used to capture the general trend, and linear regression or the Theil slope used to construct linear trend lines. If there is an interest in revealing outliers, the trend line and the prediction limits should be derived from a robust smoother and a robust estimator of the standard deviation. It would be sensible if the methods used to construct the lines are consistent throughout the assessment. To avoid cluttered graphs, some compromise may be necessary between the number of lines and confidence limits superimposed on the graph, although coloured lines might help. It will be useful to try different possibilities to find out what best suits the individual assessment.

### 6.4.2 Formal test of trend

**Objectives**

This is the core component in the package, and provides an objective test of whether there is a meaningful systematic change in the time series, assessed against some measure of the random noise in the observations. What constitutes a meaningful change will depend on the objectives of the assessment, and is a major consideration in the choice of method, discussed below.

**Type of result or information**

The output from this component will usually consist of the probability that the test statistic of the method of choice could have arisen by chance when there is no trend. If this is less than some pre-specified value (e.g., 5 %), the result is considered to be significant, i.e., the null hypothesis of no trend is rejected.

The test may be partitioned into a series of nested sub-tests, for example, partitioning the test of overall trend into tests of linear and non-linear trends. Probabilities and significance are reported for each sub-test.

**Guideline to methods, their strengths and weaknesses**

Several things must be considered when choosing a method for testing the statistical significance of a measured trend. Essentially, these are:

- Is the method relevant to the objectives of the assessment?
- Are the assumptions underlying the method valid?
- Is the method sufficiently powerful?

If the assumptions of the method include specific requirements for the distribution of the data, e.g., Normality and homogeneity of the error distribution, and these are unlikely to be met because of potential outliers, there will be a further requirement that

- the method is robust.

In the context of trend assessment, relevance means that the method is sensitive to the kinds of changes of concern in the assessment. As shown in ICES (1997), not all tests are equally effective at detecting all patterns of change. For a very focused test, this may be a disadvantage if all patterns of change are of interest, or an advantage if this focus is on patterns of interest. Robustness in the current context refers to the degree of sensitivity with regard to outliers.

Three groupings of patterns of change may be considered to be of interest:

1) linear trends,
2) monotonic trends,
3) non-monotonic trends.

Before choosing a specific method, the ACME recommends that full consideration be given to the comparisons and assessments given in the references and in Annex 1. The following table provides a very crude summary of the characteristics of four potential trend tests in terms of their power (Very good/Good/Fair/Poor) regarding these types of trends and their robustness.

| Trend Test | Power (under Normality) | | | Robust |
|---|---|---|---|---|
| | Linear | Monotonic | Non-monotonic | |
| Mann-Kendall | Very good | Fair to good | Poor | Yes |
| Compound Mann-Kendall | Good | Fair to good | Fair to good | Yes |
| Smoother | Good | Very good | Very good | Susceptible to extreme values at the start and end |
| Robust smoother | Good | ? | ? | Yes |

Here, ? denotes that the properties have not yet been studied in detail. It should also be noted that the power has been determined under the assumption of Normality.

Hence, if the purpose of the assessment is to detect monotonic trends and be unaffected by isolated extreme values, the Mann-Kendall test would be appropriate. If the purpose is to detect all trends, then the choice is between the compound Mann-Kendall test and either of the smoothers, with a final decision depending on the weight given to the other factors.

### 6.4.3 Measure of the tendency to increase or decrease

**Objectives**

One of the most consistently repeated requirements stated at assessment meetings is for some summary measurement of linear increase or decrease. Sometimes this is modified to the linear change in, e.g., the most recent seven years, but there is clearly a desire to quantify trends in this way.

**Type of result or information**

The summary statistic is the slope of some fitted straight line. The slope might also be expressed as the percentage of decrease or increase. Ideally, the standard error of the estimated slope should also be reported, although this may be misleading if there is a significant non-linear trend.

**Guideline to methods, their strengths and weaknesses**

There are many candidates for this estimated slope. If the smoother-based trend assessment is made, a natural candidate is the linear regression coefficient. If the Mann-Kendall test is made, a natural candidate is the Theil slope. However, there are many other robust regression methods that will also produce estimates of slope, and these should also be considered.

### 6.4.4 Comparison of the current level against some reference or a level in a previous year

**Objectives**

This is a somewhat specialized statistic, which may not always be required. It assumes that there exists some reference value, and that it is meaningful to compare current levels (i.e., levels at the end of the time series) against this reference. The reference value could be external, or based on an estimate of the mean at an earlier part of the time series. This might simply be the earliest year in which there is an observation (as in the Trend-y-tector), or a specified year at which some intervention occurred.

**Type of result or information**

The summary statistic might be an estimated difference together with its standard error or confidence quantile. In principle, this statistic could also be supplemented by a formal test of significance of estimated differences. However, this raises questions of the power of this comparison and how to deal with the overall significance of the multiple test. WGSAEM suggested that this could not be recommended at that stage.

**Guideline to methods, their strengths and weaknesses**

This statistic will almost certainly be derived from the fitted smoother. Annex 1, Part 3 compares the performance of the smoother and an estimate based on the Theil slope. This shows that with monotonic trends the performance of the Theil-slope method was poor. Similarly, the performance of the smoother was poor in the presence of outliers. This is an area where some further assessment of methods is desirable.

*Need for further research or additional data*

Further research is needed:

- to specify appropriate adjustment techniques for loads and to explore the use of monthly data;

- to develop and assess robust smoother methods.

*Recommendations*

Based on the above material, with regard to temporal trend assessments in general, and the assessment of inputs in particular, the ACME recommends the procedures described below.

For temporal trend assessments of annual indices, an appropriate package of statistical tools should separate four components, with some flexibility as to how these components are implemented. Effectively, the components would provide:

a) a scatterplot of index against year with a superimposed line capturing the general trend. This plot could be superimposed with reference lines, forecasts, etc., as appropriate;
b) the result of a single formal test of the temporal trend using an appropriate method;
c) an appropriate measure of the linear trend; and
d) an appropriate measure of the difference between the estimated mean in the current year against a reference value or the estimated mean level in some previous year.

For the assessment of temporal trends in an annual index of contaminant inputs (e.g., annual unadjusted load, annual adjusted load, etc.), OSPAR should decide between two specific implementations of this package of statistical tools. These implementations should satisfy the immediate requirements of OSPAR, and also ensure that the characteristics of the components of these implementations are well documented and understood.

Implementation 1

The following implementation should be used if the objective of OSPAR is to detect underlying monotonic trends excluding individual extreme points:

1) A scatterplot of the annual index against year with a superimposed seven-year-window LOESS smoother. Confidence limits for the smoother, a linear trend based on the Theil slope, and any other reference lines may be added, possibly using colour.
2) A formal test of trend using the Mann-Kendall test.
3) The Theil slope.
4) The difference between the estimated mean in the final year and the estimated mean in the first year (1991) derived from the smoother. If an upper confidence quantile is reported, the standard error is derived from the residual standard deviation from the smoother.

Implementation 2

The following implementation should be used if the objective of OSPAR is to detect underlying both monotonic and non-monotonic trends:

1) A scatterplot of the annual index against year with a superimposed seven-year-window LOESS smoother. Confidence limits for the smoother, a linear trend based on the Theil slope, and any other reference lines may be added, possibly using colour.
2) A formal test using the smoother, with a partition into linear and non-linear trends.
3) The linear regression coefficient.
4) The difference between the estimated mean in the final year and the estimated mean in the first year (1991) derived from the smoother. If an upper confidence quantile is reported, the standard error is derived from the residual standard deviation from the smoother.

Users of any implementations should be prepared to review the performance after a period of use. The implementation can then be revised and fined-tuned to satisfy specific needs more closely, and to incorporate new and improved methodologies as they are developed.

With regard to the adjustment of inputs, the ACME recommends that for temporal trend assessments of loads, an inclusion of an appropriate adjustment step prior to the application of the trend tests package should be considered. Adjustment aims to compensate for the effect of irregular fluctuations induced by varying flow rates and precipitation. Preliminary analyses are promising and increased power of trend detection is to be expected. However, the development of an appropriate adjustment procedure may well be revised as more data and experience accumulate.

**Reference**

ICES. 1997. Report of the ICES Advisory Committee on the Marine Environment, 1997. ICES Cooperative Research Report, 222: 26–29.

# 7 QUALITY ASSURANCE PROCEDURES AND INTERCOMPARISON EXERCISES

## 7.1 Quality Assurance of Biological Measurements in the Baltic Sea

### 7.1.1 Progress on QA for biological methods in the HELCOM COMBINE Programme

*Request*

Item 2 of the 1999 requests from the Helsinki Commission: to coordinate quality assurance activities on biological and chemical measurements in the Baltic Sea 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 1999 report of the ICES/HELCOM Steering Group on Quality Assurance of Biological Measurements in the Baltic Sea (SGQAB) and ACME deliberations.

*Status/background information*

The introduction of quality assurance (QA) procedures into the biological measurements in the HELCOM Cooperative Monitoring in the Baltic Marine Environment (COMBINE) Programme has continued since the establishment of SGQAB in 1992. The main emphasis has been to harmonize the QA procedures in such a way that they are applicable for both chemical and biological analyses and, when needed, to adopt specific procedures for biological measurements. The following progress has been made since the 1998 meeting of ACME:

- The revision of the biological QA and methodological parts of the COMBINE Manual has been continued.
- The ICES guidelines for primary production measurements (Colijn and Edler, in prep.) was adopted, with the recommendation that it be included into the COMBINE Manual.
- The status of the ICES Biological Data Reporting Format and data entry program was discussed with ICES personnel and changes were proposed.
- The work of taxonomic training courses on phytoplankton and benthos has been continued.
- The taxonomic Checklist of Baltic Sea Phytoplankton Species was essentially completed by G. Hällfors and will be ready for printing in 2000.
- A proposal to arrange annual phytoplankton intercalibration exercises was made by the Alg@line Project of the Finnish Institute of Marine Research.

The proposal to change the sampling depth for phytoplankton analysis, made at the 1997 ICES/HELCOM Workshop/Training Course on Phytoplankton, was discussed by SGQAB. This proposal, to change the sampling depth from 0–10 m to 0–20 m, was made to provide a more complete description of species composition. However, a disadvantage is that it may cause a break in relevant time series of data, making the new data impossible to compare with the data obtained using the earlier sampling depth guidelines. Nonetheless, SGQAB noted that long-term data sets are in many cases poor, owing to a low frequency of sampling. Thus, SGQAB decided to address this question to the ACME (as well as to the HELCOM Environment Committee Working Group on Monitoring and Assessment (EC MON)) in order to ascertain the extent to which scientific knowledge and accuracy should affect monitoring policy and what approach should be used for assessing long-term trends.

In response, the ACME emphasized that, according to the accepted QA procedures, prior to making changes in the sampling procedures or analysis methods, possible effects on the results should be documented and evaluated. Because no detailed supporting material on this subject was provided by the ICES/HELCOM Workshop/Training Course on Phytoplankton, the ACME can give no recommendation concerning the proposed change in sampling depth for phytoplankton. This material should be prepared and submitted via SGQAB when available.

*Need for further research or additional data*

As pointed out by ACME in 1998, the lack of biological reference materials and regular intercalibration exercises are still a matter of concern. However, the QUASIMEME and BEQUALM projects will include some biological analyses in their programmes, which will be helpful in the future.

The updating of taxonomic checklists of species is a continuous task in order to guarantee the quality of the long-term data sets. The continuation of this work should be ensured in the future. It has been found possible to improve the identification of species by providing identification material on the Internet. The production of such material should be organized and harmonized. The Working Group on Phytoplankton Ecology (WGPE) could have a coordinating role in this process.

Regular ring tests for all laboratories that submit data for the COMBINE Programme should be organized. The five-year frequency, recommended earlier by ACME, seems to be too low in order to improve the quality of the data. Therefore, at least annual ring tests should be organized. In order to avoid overlapping, and to harmonize national and international ring tests, coordination is necessary.

The ACME agreed that the ICES/HELCOM Steering Group on Quality Assurance of Biological Measurements in the Baltic Sea should continue its work. Proper QA procedures are a prerequisite for the COMBINE Programme. The ACME requested SGQAB to develop regular ring tests for all biological core and main variables in the COMBINE Programme, including the open sea and coastal components.

The ACME further requested the Phytoplankton Training Course to test the effect of the proposed change in sampling depth for phytoplankton analysis and report the results for further evaluation by SGQAB.

The ACME took note of the need for biological data reporting formats and agreed that high priority should be given to their completion by the ICES Environmental Data Centre.

Updated taxonomic species lists are a continuous task, also for other types of organisms than phytoplankton. Identification material for all relevant species should be made available on the Internet. The ACME requested the relevant Working Groups, such as WGPE, the Working Group on Zooplankton Ecology (WGZE), and the Benthos Ecology Working Group (BEWG), to coordinate this work.

*Recommendations*

ICES recommends that the QA and analysis sections prepared for the COMBINE Manual, that have been adopted by SGQAB and endorsed by ACME, should be forwarded to HELCOM, with a proposal for their incorporation in the COMBINE Manual.

*Additional Comments*

ICES expressed its gratitude to Mr Lars Hernroth (Sweden), the outgoing Chair of SGQAB, and acknowledged his important contribution to the work of the Steering Group.

**Reference**

Colijn, F., and Edler, L. In prep. Working manual and supporting papers on the use of a standardized incubator technique in primary production measurements. Manuscript to be submitted to the ICES Techniques in Marine Environmental Sciences series.

### 7.1.2 Results of the ICES/HELCOM Workshop/Training Course on Phytoplankton

*Request*

Item 2 of the 1999 requests from the Helsinki Commission.

*Source of the information presented*

The 1998 report of the ICES/HELCOM Workshop/Training Course on Phytoplankton (WKPHYT), the 1999 report of the ICES/HELCOM Steering Group on Quality Assurance of Biological Measurements in the Baltic Sea (SGQAB), and ACME deliberations.

*Status/background information*

The ACME noted that, at its Nineteenth Meeting, the Helsinki Commission re-established the project 'Quality Assurance of Phytoplankton Monitoring in the Baltic Sea' (HELCOM, 1998) for 1998–2000, thus ensuring that training courses on phytoplankton species identification and microscope counting techniques could continue to be organized on an annual basis. In 1998, WKPHYT met in Klaipeda, Lithuania in October; the course was composed of two parts: training on cyanobacteria (Cyanoprocaryota) and training on flagellates (Prymensiophycae, Chrysophyceae, Cryptophyceae, Prasinophyceae). In addition, other items, related to the COMBINE Manual, were discussed as follows:

- suggestions for changes to the Manual were discussed and proposed for further consideration by SGQAB and the ACME, and adoption by the HELCOM Environment Committee Working Group on Monitoring and Assessment (EC MON);
- progress in the development of a new phytoplankton taxonomic checklist was reviewed and it was concluded that continuation of this work is essential;
- improvements to the HELCOM phytoplankton counting program were considered;
- activities and decisions of the biomass sub-group were reported;
- the proposed change in sampling depth from 0–10 m to 0–20 m, in order to obtain more realistic species composition and biomass values, was reaffirmed (see also Section 7.1.1, above).

It was noted that the next meeting of WKPHYT is scheduled to take place in Abisko, Sweden, in early September 1999. The taxonomic training will concentrate on Chlorophyceae and *Chaetoceros* species.

*Need for further research or additional data*

The ACME agreed that the organization of taxonomic training courses and the expert evaluation of working manuals conducted by WKPHYT are parts of the continuous development of QA. Intercomparisons, intercalibrations, or ring tests on the analysis of phytoplankton samples should be arranged annually. Additionally, there is still a lack of reference materials for phytoplankton.

The ACME requested WKPHYT to assist in organizing annual ring tests on phytoplankton analysis.

Referring to Section 7.1.1, above, the ACME also requested WKPHYT to submit proper background data on the effects of changes in sampling depth on long-term data sets on phytoplankton.

*Recommendations*

ICES emphasizes that taxonomic expertise is essential for phytoplankton monitoring. The regular and systematic organization of training courses and the participation of expert staff from all laboratories submitting monitoring data to the HELCOM database are recommended.

*Additional comments*

ICES expresses its gratitude to the lecturers in the Training Course: Ms Gertrude Cronberg (University of Lund), for training on cyanobacteria, and Ms Seija Hällfors and Mr Guy Hällfors (Finnish Institute of Marine Research), for training on flagellates.

**Reference**

HELCOM. 1998. Report of the Nineteenth Meeting of the Helsinki Commission. HELCOM 19/98, 15/1.

## 7.2 Quality Assurance of Biological Measurements in the OSPAR Area

*Request*

Item 2.1 on the 1999 Work Programme from the OSPAR Commission:

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

a) the development of quality assurance procedures;

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

c) the preparation of appropriate taxonomic lists of species.

This work should cover the biological parameters within the eutrophication monitoring guidelines, namely: chlorophyll-*a*, phytoplankton, macrozoobenthos and macrophytobenthos. This is a fairly long-term programme (about five years) requiring the participation of scientists and technicians carrying out relevant analyses for this monitoring in laboratories in OSPAR Contracting Parties. Good cooperation should be ensured with the ICES/HELCOM Steering Group on Quality Assurance of Biological Measurements in the Baltic Sea.

*Source of the information presented*

The 1999 report of the ICES/OSPAR Steering Group on Quality Assurance of Biological Measurements related to Eutrophication Effects (SGQAE) and ACME deliberations.

*Status/background information*

The ACME reviewed the SGQAE report, which presents information on eutrophication-related monitoring programmes developed in several OSPAR and ICES countries and the QA activities associated with these programmes. Activities in the following countries were described: Belgium, Germany, the Netherlands, Norway, Sweden, and the UK. In addition, QA issues arising from discussions within other ICES or international groups were presented and discussed.

The ACME noted the following main topics that were considered:

1) The development of QA procedures for the measurement of primary production, chlorophyll *a* (in cooperation with the Marine Chemistry Working Group (MCWG) (see Section 7.3, below)), and phytoplankton species by the Working Group on Phytoplankton Ecology (WGPE).

2) The need for the development of QA approaches to cover issues of survey objectives and sampling design, which has been agreed by the Benthos Ecology Working Group (BEWG). BEWG wishes to establish the degree of variability in current practices in order to allow recommendations to be prepared.

3) In the joint session between SGQAE and SGQAB, harmonization of the guidelines developed for the HELCOM COMBINE Programme and for OSPAR and ICES was discussed, as well as a proposal for taxonomic training courses and annual ring tests on phytoplankton analysis. Guidelines for phytobenthos monitoring, produced for HELCOM, were reviewed.

4) The Guidelines on QA for Chemical Measurements in the Baltic Sea, prepared by the ICES/HELCOM Steering Group on Quality Assurance of Chemical Measurements in the Baltic Sea (SGQAC), have been

recommended for publication in the *ICES Techniques in Marine Environmental Sciences* series.

5) The EU-funded BEQUALM project includes work on the improvement of chlorophyll *a* measurements, phytoplankton species identification, and macrozoobenthos QA procedures (see also Section 7.4, below). A ring test on chlorophyll *a* measurements has been planned (together with QUASIMEME).

The ACME noted that SGQAE also discussed the problem of evaluating the acceptability of data; criteria were presented that are used within the UK National Marine Biological Analytical Quality Control Scheme (NMBAQC) for ensuring that macrobenthos data are of consistent and acceptable quality.

Some of the main outputs from SGQAE include:

1) Preparation of a draft document on guidelines for QA of biological measurements under OSPAR/ICES, including: i) an exhaustive summary of the general QA system for application to aquatic environmental sciences, and ii) critical QA factors for specific parameters to be monitored, i.e., chlorophyll *a*, phytoplankton, macrozoobenthos, and macrophytobenthos. The basis for this document was the corresponding HELCOM COMBINE Manual Part B (the general QA guidelines prepared originally by SGQAC and later modified by SGQAB), a Guidance Document on QA in Environmental Monitoring published by the Nordic Council of Ministers (NCM, 1997), and other relevant documents.

2) A review of the draft data reporting format for biological parameters produced by the ICES Environmental Data Centre. Several additions and changes to the reporting format were suggested.

A large part of the Guidelines for QA of biological measurements is devoted to general aspects of the QA system that should be applied to any environmental study. These rules could constitute General ICES/OSPAR QA Guidelines that either could be issued separately or that may be published as the first part of the specific QA guidelines.

The ACME agreed that it is essential that methodologies recommended in QA guidelines be in accordance with methods currently accepted by ICES Working Groups (e.g., chlorophyll measurements). Unless justified, recommendations should be identical for OSPAR and HELCOM and should comply, in this example, with the method elaborated by WGPE and MCWG.

Members of SGQAE were requested to provide examples of good practice for inclusion in the QA Guidelines document. These examples should not be included in the body of the general text, but rather for better readability should be presented together with the critical QA factors listed for each of the specific parameters. In that way, analysts would find all specific information grouped together within the document.

In conclusion, the ACME appreciated the well-documented outputs from SGQAE, and recommended that the group finalize the draft guidelines for QA procedures, including the addition of specific examples.

**Reference**

NCM. 1997. Quality assurance in environmental monitoring. TemaNord 1997:591. Nordic Council of Ministers, Copenhagen, Denmark. 87 pp.

## 7.3 Standard Technique and Quality Assurance Procedures for Chlorophyll Determinations in Sea Water

*Request*

There is no specific request; this is part of the continuing work of ICES on quality assurance of marine monitoring, but is also of relevance to monitoring under OSPAR and HELCOM.

*Source of the information presented*

The 1999 reports of the Marine Chemistry Working Group (MCWG) and the Working Group on Phytoplankton Ecology (WGPE), and ACME deliberations.

*Status/background information*

Chlorophyll *a* is included, as a biomass marker, in monitoring programmes aimed at investigating eutrophication effects. There has been a requirement in ICES to identify a standard method for chlorophyll *a* determination in order to be able to compile a more useful and comparable database on chlorophyll. It is a requirement that a monitoring method such as this is practical and easy to conduct. At the same time, it is important to be aware of health risks (extraction solvent) and costs.

At its 1999 meeting, WGPE discussed a proposal for a standard procedure for chlorophyll *a* determination based on spectrophotometric procedures. This standard procedure had been prepared in consultation with MCWG. WGPE pointed out that other methods may be used, but in that case it must be shown that the other method agrees with the standard method and any discrepancies must be identified and justified. The ACME reviewed the proposed Standard Procedure for the Measurement of Chlorophyll *a* in the ICES area and accepted it for use in ICES.

Quality assurance procedures must be followed with the standard method. In 1999, MCWG reviewed and accepted the document 'Overview and Recommendations for the Determination of Chlorophyll *a* by Spectroscopic Methods'. The ACME reviewed a revised version of this document and accepted it as a complement to the description of the standard procedure described above.

Although the ACME originally decided to annex these two documents to its report, it was subsequently agreed that they should be merged into one overall document and published in the *ICES Techniques in Marine Environmental Sciences* series (see Aminot and Rey, in prep.).

*Recommendations*

ICES recommends that the Standard Procedure for the Measurement of Chlorophyll *a*, contained in Aminot and Rey (in prep.), be used in the ICES area and emphasizes that appropriate quality assurance procedures must be followed in phytopigment studies.

**Reference**


Aminot, A., and Rey, F. In prep. Standard procedure for the determination of chlorophyll *a* by spectroscopic methods. Manuscript submitted for publication in ICES Techniques in Marine Environmental Sciences.


### 7.4 Quality Assurance Procedures for Biological Effects Techniques, including Fish Diseases

*Request*

There is no specific request; this is part of continuing ICES work on quality control procedures.

*Source of the information presented*

The 1999 reports of the Working Group on Biological Effects of Contaminants (WGBEC) and the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO), and ACME deliberations.

*Status/background information*

All biological effects monitoring techniques required for use under the OSPAR Joint Assessment and Monitoring Programme (JAMP), with the exception of those for measuring oxidative stress, are the subject of an EU-funded programme known as BEQUALM (Biological Effects Quality Assurance in Marine Monitoring), which is setting up a quality control scheme for these techniques. The techniques include sediment and water bioassays, EROD induction, metallothionein induction, δ-aminolevulinic acid (ALA-D) inhibition, lysosomal stability, benthic community analysis, fish liver pathology, and phytoplankton assemblage analysis. If successful, the BEQUALM programme is intended to become self-financing in 2001.

BEQUALM has been running since November 1998 and practical activities are now starting under the leadership of a number of expert laboratories. The activities consist of a variety of workshops to agree on methodology and, in some cases, to provide training, the development and implementation of intercalibration methods, and the establishment of performance assessment programmes. All activities are being conducted with a number of participating laboratories in many European countries. As an example, a workshop on the histopathology of fish liver lesions is scheduled to take place at the CEFAS Laboratory in Weymouth, UK, in October 1999. Lists of participants have already been assembled, but it is not too late for additional laboratories to join the BEQUALM programme. Interested persons can consult http://www.cefas.co.uk/bequalm (the BEQUALM website)—which includes full details of the work packages as well as contact names and addresses—or contact the project leader, Dr P. Matthiessen, directly at the CEFAS Laboratory (p.matthiessen@cefas.co.uk) in Burnham, UK.

The ACME took note of the above information on progress in the BEQUALM programme and endorsed the development of quality control procedures for all biological effects monitoring techniques.

### 7.5 Quality Assurance of Chemical Measurements in the Baltic Sea

*Request*

Item 2 of the 1999 requests from the Helsinki Commission: to coordinate quality assurance activities on biological and chemical measurements in the Baltic Sea 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 1999 report of the ICES/HELCOM Steering Group on Quality Assurance of Chemical Measurements in the Baltic Sea (SGQAC) and ACME deliberations.

*Status/background information*

The ACME reviewed the work of the ICES/HELCOM Steering Group on Quality Assurance of Chemical Measurements in the Baltic Sea (SGQAC) and noted that the Guidelines on Quality Assurance of Chemical Measurements in the Baltic Sea, as prepared by SGQAC and accepted by ACME, have been incorporated in the Manual for Marine Monitoring in the COMBINE Programme of HELCOM. They can be found at http://www.helcom.fi/manual2/contents.html.

Further progress by SGQAC in the development of additional Technical Annexes to the Guidelines was noted, as follows:

1) Technical Notes were completed on quality assurance procedures for the determination of the following hydrochemical parameters: dissolved oxygen, hydrogen sulphide, pH, and alkalinity;
2) further progress was made on QA for sampling and sample handling;
3) progress was reviewed on the preparation of technical notes on the analyses of chlorinated biphenyls (CBs) and organochlorine pesticides (OCPs) in sea water;
4) progress was reviewed on the preparation of technical notes on the determination of polycyclic aromatic hydrocarbons (PAHs) in sea water and biota;
5) a paper on measurement uncertainty in chemical analysis was reviewed, for further consideration in 2000.

The ACME was informed that technical notes on quality assurance procedures for the determination of temperature and salinity in sea water were also under preparation. It is anticipated that these technical notes will be finalized in 2000, in cooperation with the Working Group on Shelf Seas Oceanography (WGSSO) and the Working Group on Marine Data Management (WGMDM).

The ACME accepted the technical annexes that have been completed, namely:

- Technical notes on the determination of dissolved oxygen in sea water;
- Technical notes on the determination of hydrogen sulphide;
- Technical notes on pH measurement in sea water;
- Technical notes on measurement of total alkalinity.

These technical annexes will be transmitted to HELCOM for inclusion in the COMBINE Manual. It was felt that it would be appropriate to combine the four above-mentioned technical notes, together with the technical notes on temperature and salinity when they have been completed, into one overall technical annex on hydrochemical parameters, to provide a more user-friendly document.

It was noted that the information in the present technical annexes to the Guidelines will be updated, when appropriate, according to the new developments in methodology, and new technical annexes on additional topics will be prepared as required.

The ACME noted a summary of an evaluation report on the results of a questionnaire on laboratory performance that had been distributed to HELCOM laboratories in 1998. These results indicated that the laboratories, in general, are capable of fulfilling the requirements stated in the COMBINE Manual for measurements of temperature and salinity. With regard to nutrients, it was felt that the limits of detection values set in the COMBINE Manual are unrealistically low and, therefore, are barely met by the laboratories. SGQAC was concerned that the laboratories might devote too much effort to lowering their detection limits, which is unnecessary with regard to trend monitoring of winter nutrient concentrations. Since a QUASIMEME evaluation based on $Z<2$ is satisfactory (see Section 5.4, above), this level of analytical performance should also be recommended for HELCOM laboratories.

It was noted that for many parameters the evaluation was severely hampered by the lack of clear analytical requirements in the COMBINE Manual. It was also difficult, from the general goals of the programme, to objectively calculate the necessary analytical performance. It must also be kept in mind that the required analytical performance is strongly linked to other influences, including sea dynamics, sampling frequency or spatial coverage. Accordingly, the ACME recommends that HELCOM develop more specific analytical requirements for all parameters in the COMBINE programme.

The ACME noted and supported the plans for the Second ICES/HELCOM Workshop on Quality Assurance of Chemical Analytical Procedures for the COMBINE and PLC-4 Programmes, scheduled to take place in Helsinki, Finland on 21–23 October 1999. This Workshop will cover external and internal quality assurance measures for the monitoring of nutrients/hydrographic parameters, trace metals, and organic contaminants. In this connection, the ACME advised continuing work on physical/hydrochemical parameters and cofactors.

*Need for further research or additional data*

The ACME noted that the Guidelines for Quality Assurance of Chemical Measurements in the Baltic Sea and the associated technical annexes will be updated in line with methodological progress and the needs of the HELCOM COMBINE monitoring programme.

*Recommendations*

ICES recommends that the technical notes on the hydrochemical parameters dissolved oxygen, hydrogen sulphide, pH, and alkalinity be forwarded to HELCOM for inclusion in the Guidelines on Quality Assurance of Chemical Measurements in the Baltic Sea and use in the COMBINE Monitoring Programme

ICES recommends that HELCOM COMBINE laboratories regularly participate in external quality assurance schemes in order to ensure the accuracy and comparability of their data.

## 7.6 Developments within QUASIMEME and QUASH

*Request*

There is no specific request; the ACME addressed this item because of the long-standing ICES involvement in quality assurance matters.

*Source of the information presented*

The 1999 reports of the Marine Chemistry Working Group (MCWG) and the Working Group on Marine Sediments in Relation to Pollution (WGMS), and ACME deliberations.

*Status/background information*

Reports on the progress of QUASIMEME, and more recently also on QUASH, have been on the ACME agenda regularly for several years.

It is now six years since the QUASIMEME programme began in 1993. It has meanwhile changed from an EU pilot project to a Laboratory Quality Assurance Performance programme supported by its participating members. More than 100 laboratories worldwide have joined the scheme. ICES is represented on the Advisory Board, along with representatives from OSPAR and HELCOM. The active feedback from the participants and the Advisory Board is a key element in the development and improvement of the programme.

The main activities and developments in the 1998/1999 QUASIMEME programme have been:

- the conduct of Training Workshops;
- launch of a Windows-based data transfer system;
- the conduct of an extended development programme, including establishing links with biological effects and the EU BEQUALM project (see Section 7.4, above);
- the preparation of improved test materials.

The development exercises have been designed to provide a series of laboratory studies followed by a workshop to evaluate the exercise, provide training, and plan for subsequent studies.

There have been three workshops during 1998/1999, covering the following areas:

1) Chemical determination of organotin compounds, a workshop held at the Institute for Environmental Studies (IVM), Free University, Amsterdam;
2) Determination of PAH metabolites, a joint workshop with the IVM, Free University, Amsterdam, and the Rogaland Research Institute, Norway, in the EU Standards, Measurements and Testing Programme;
3) Chlorobornanes (CHBs, chlorinated camphenes or toxaphene) and their measurements in biota, a workshop held at the University of Basel.

The Windows-based data transfer system has made it possible to provide each participant with a database with their own performance results which can be displayed, printed or passed to a third party as a part of the quality assurance of their environmental monitoring data.

New programmes include, for example, the development of test material comprising marine snails for the determination of imposex, a progression of the organotin programme, and test materials on chlorophyll in solution and on filter papers. Both the chlorophyll programme as well as the imposex programme will be continued, the latter in cooperation with the new EU project BEQUALM, which will be involved in establishing methods of quality assurance for a wide range of biological effects methods (see Section 7.4, above).

The QUASH project started in October 1996 and is due to end in March 2000. Interlaboratory studies focusing on sampling and sample handling of sea water, biota, and sediments have been carried out in all of the QUASH work packages, and practical workshops have been held to discuss the results from these exercises. A second phase of the interlaboratory studies for the biota and sediment handling work packages is presently under way. The outcome of these exercises will, e.g., provide outstanding and long-awaited contributions to the discussions on normalization of contaminant concentrations in sediment samples, as well as giving practical experience in several normalization techniques including sieving. WGMS has considered these results to be so essential that it is necessary to await them before the normalization annex of the sediment monitoring guidelines, drafted last year, can be finalized.

A great deal of work on QUASH has been done at the international level, but there have also been a number of initiatives at the national level in a number of countries.

A final workshop is scheduled to be held in October 1999 in the Netherlands, where the QUASH project will be reviewed and information on all the work packages, including workshop reports, will be presented.

*Additional comments*

The ACME will continue to follow the progress in QUASIMEME II and QUASH.

**Reference**

QUASIMEME. 1999. QUASIMEME Bulletin, Issue No. 6. Marine Laboratory, Aberdeen, UK. For further

information consult the QUASIMEME website at http://www.quasimeme.marlab.ac.uk/.

## 7.7 Laboratory Quality Assurance Database Systems

*Request*

There is no specific request; this is part of the continuing ICES work on quality assurance.

*Source of the information presented*

The 1999 report of the Marine Chemistry Working Group (MCWG) and ACME deliberations.

*Status/background information*

Knowing the quality of the data in a database is essential to ensure an appropriate use of the data. This requires both the production of high quality analytical data as well as the storage of quality assurance (QA) data in a database with linkages to the analytical results of the samples. Both topics were discussed by MCWG.

A QA system for use in a laboratory involved in marine monitoring was reviewed, based on the experience with the QA system used in the German Marine Monitoring Programme. Due to the involvement of over twenty laboratories in the German monitoring programmes, national coordination of the procedures for the QA of data is required, in order to be able to compare the data with some confidence. The QA procedure used is based on three main components:

1) validation of analytical methods;
2) internal quality control (including the use of certified reference materials (CRMs) or in-house reference materials and the plotting of quality control (QC) charts);
3) participation in interlaboratory performance studies.

There was general agreement that the three main components of the German QA procedures also comprised the basic requirements for the QA procedures in most laboratories. It was, however, pointed out that the use of CRMs does not always provide adequate information on the ability of the laboratories to measure real samples. The reported standard deviation on the analysis of CRMs is always much smaller than the observed relative errors in intercomparison exercises. The CRMs used must match the samples with regard to sample matrix and contaminant levels to be informative.

The reporting of QA data together with monitoring data has been the subject of an ongoing discussion in MCWG. The discussion has focused on the differences between the QA information produced by laboratories for their national programmes and the information that is normally sent by them to ICES. More QA data are usually reported for national programmes than are required by ICES, including different CRMs, QC charts, and the results of laboratory performance studies. The procedures used in different countries varied, however, e.g., due to the fact that some countries' laboratories report directly to ICES according to the format used by ICES, whereas others report to a national data centre.

The method used by Ireland was also reviewed. It involves the use of an ACCESS database for entering the monitoring data and a linked ACCESS database for the QA data. Each sample in the monitoring database is linked to the QA database through an identifying code. This method also provides a rapid means of transferring data to the ICES database. It was stressed that there is a need to update the reporting format for QA data used by ICES if the QA data are to be used in the assessment of monitoring data. It was also stressed that guidelines on how QA data are to be used in the quality assessment of monitoring data need to be established (see also Section 5.6.4, above). Reference was made to an approach to quality assessment recommended by Dobson *et al.* (1999). The QA evaluation of monitoring data should not take place at the last moment, but should be ready before the statistical analysis and interpretation of monitoring data are conducted.

It was also emphasized that data transfer may be a source of error. The data transfer procedures should be checked and should, in fact, have their own QA procedure.

The ACME noted that MCWG will continue work on this topic at its meeting in 2000.

**Reference**

Dobson, J., Gardner, M., Miller, B., Jessep, M., and Toft, R. 1999. *An approach to the assessment of the quality of environmental monitoring data*. Journal of Environmental Monitoring, 1: 91–95.

# 8 ENVIRONMENTAL STUDIES

## 8.1 Effects of Extraction of Marine Sand and Gravel on the Marine Ecosystem

*Request*

There is no specific request; this is part of continuing ICES work on ecosystem effects of marine aggregate extraction.

*Source of the information presented*

The 1999 report of the Working Group on the Effects of Extraction of Marine Sediments on the Marine Ecosystem (WGEXT) and ACME deliberations.

*Status/background information*

The ACME reviewed and accepted several sections of the report of the Working Group on the Effects of Extraction of Marine Sediments on the Marine Ecosystem (WGEXT) containing information and discussions on the effects of marine sand and gravel extraction on marine ecosystems, including quantities of material extracted, impacts on biota, and the effect of turbidity caused by dredging. Approaches to environmental impact assessment were also reviewed, as well as techniques for seabed characterization.

### 8.1.1 Review of national marine aggregate extraction activities

The status of marine extraction and dredging activities in ICES Member Countries, as reported to WGEXT, is provided in the following paragraphs.

**Belgium**

In 1998, 1 391 632 $m^3$, mainly of sand, was extracted off the coast of Belgium.

**Canada**

Most aggregate extraction in Canada has been limited to the dredging of harbours and channels for navigation or construction purposes. In 1995, for the Atlantic Provinces of Canada, twelve projects resulted in the dredging and dumping of 331 000 $m^3$; in 1996, fifteen projects dumped 844 000 $m^3$; and in 1997, the most recent year with final figures, thirteen projects resulted in the dumping of 284 000 $m^3$.

As a result of a conference of Provincial mines ministers in 1998, the Canadian government decided to investigate the development of legislation to permit and control marine mining. Previous attempts in the 1980s were unsuccessful at implementing a marine mining regime.

Canada has expanded the collection of multibeam bathymetric data in the coastal zone with the acquisition of Simrad EM 3000 systems, mounted on coastal survey vessels, for operations from the beach to depths of 100 m with decimetre resolution. Multibeam surveys are now routinely used by the marine geological community and are generally conducted before seismic, side-scan and sampling surveys.

**Denmark**

The extraction of marine sand and gravel represents 10–13 % of the total production of materials for construction and reclamation in Denmark. The dredging of sand fill for land reclamation has increased markedly over the past ten years, caused by several large construction works in coastal areas.

From 1989 to 1993, more than $9 \times 10^6$ $m^3$ of sand fill and till were dredged for the construction of the Great Belt bridge and tunnel project. During the construction of the fixed link between Denmark and Sweden, $1.3 \times 10^6$ $m^3$ has been dredged, with a spill of only 2.8 %. In the same period, $7 \times 10^6$ $m^3$ of dredged materials of glacial till and limestone has been reused for reclamation and as hydraulic fill in ramps for the bridge and tunnel.

A major enlargement of the harbour of Århus is expected to require more than $5 \times 10^6$ $m^3$ of sand fill in the next two years. The construction works started in the autumn of 1998. Thus far, $1.3 \times 10^6$ $m^3$ has been dredged from two areas in Århus Bight. The spill from the dredging operations has been 3.7 %.

The consumption of sand for beach replenishment on the west coast of Jutland has shown a pronounced increase, from 40 000 $m^3$ in 1980 to more than $3.5 \times 10^6$ $m^3$ in 1998.

In 1996, the Danish National Forest and Nature Agency commissioned the Geological Survey to evaluate the total reserve volume of sand and gravel in Danish waters. A total of $4500 \times 10^6$ $m^3$ has been identified so far, $3500 \times 10^6$ $m^3$ of which should be available for dredging. Most of the materials are sand for fill and construction, with only limited resources of coarse aggregates available. The present knowledge of resources and environmental conditions in the North Sea and the Baltic Sea is very incomplete. However, it is expected that there are large resources in unmapped areas.

**France**

Marine aggregate extraction has been stable over recent years, with an annual production of around $3.5 \times 10^6$ tonnes.

## Germany

North Sea

The largest amount of sediment extraction is derived from maintenance dredging within estuaries and supporting waterways, accounting for between $45 \times 10^6$ and $55 \times 10^6$ tonnes per year. The extraction of sand for beach nourishment purposes is also undertaken and the total amount of marine aggregate (sand) dredged in 1998 was $11.6 \times 10^6$ $m^3$.

Baltic Sea

The situation in Schleswig-Holstein is as previously reported, i.e., no extraction of marine sediments has taken place or is currently planned. In the near future, a map showing all actual extraction areas on the coastal shelf of Mecklenburg-Vorpommern will be published by the Federal Agency for Nature Conservation. There are twenty extraction sites licensed for coastal defense purposes. These sites are not permanently exploited, but are used periodically for coastal defense projects in the respective regions. A total of fourteen extraction sites are licensed for commercial use. The total amount of material extracted in 1997 was $2.269 \times 10^6$ $m^3$.

## Ireland

The commercial dredging activities in Ireland for sand and gravel are not well developed; accordingly, only two cases were reported:

1) in Cork Harbour, 800 000 tonnes of aggregate were dredged to use as backfill in a cross-harbour road tunnel;
2) approximately 5000 tonnes of *Lithothamnium* sp. (calcareous algae) were dredged in Bantry Bay, and processed and sold as various high added-value products to the agricultural industry.

## The Netherlands

Sand extraction in 1998

The amount of sand extracted from the Dutch sector of the North Sea was $21.5 \times 10^6$ $m^3$ in 1998. About $0.073 \times 10^6$ $m^3$ has been dredged on the Danish shelf for a gravity foundation and dumped on the Dutch shelf. The main applications of extracted sand are for the beach nourishment programme and for land uses. In 1998, approximately $7.4 \times 10^6$ $m^3$ was used for beach nourishment and about $14.1 \times 10^6$ $m^3$ was used mainly for landfill, with a small part of this quantity used for the construction industry in Belgium and the southwestern part of the Netherlands.

Gravel extraction in 1998

No extraction of gravel took place in the Dutch part of the North Sea during 1998. Due to the Dutch policy on the extraction of surface minerals, as described in the Structure Plan for Surface Minerals, extraction on the Claever Bank will not be permitted before the termination of gravel extraction carried out in conjunction with the lowering of the winter bed of the Maas River. This policy is influenced by a major Delta plan project for the southeastern part of the Netherlands. The effects of this plan will lead to the production of large quantities of gravel over a short period of time, due to be completed by 2005. The total available quantities will increase from $35 \times 10^6$ to $60 \times 10^6$ tonnes due to these works.

Shell extraction in 1998

On the basis of the National Policy plan and the environmental impact assessment (EIA) for shell extraction, there are maximum permissible amounts defined from 1999 onwards. The total amounts of shell extracted in 1998 were: 107 993 $m^3$ in the Wadden Sea; 52 007 $m^3$ in sea inlets; 16 600 $m^3$ in the Western Scheldt; and 55 285 $m^3$ in the Voordelta.

## Norway

As has been the case for the past five years, no sand or gravel extraction took place on the Norwegian shelf in 1999. However, carbonate sand has been extracted along the west coast, mainly in the counties of Rogaland and Hordaland. There has been a decrease in volume, from on average 100 000–150 000 tonnes per year between 1990 and 1995, to 78 000 tonnes in 1996, 87 000 tonnes in 1997, and 52 000 tonnes in 1998.

## Poland

Gravel extraction

Only very limited-scale exploitation has been carried out on the Slupsk Bank. In 1997, about 1900 tonnes and, in 1998, about 2000 tonnes of gravel were extracted.

Sand extraction

Sand is extracted in Poland for coastal defense purposes only. In 1997, a total of 443 000 $m^3$ was extracted from ports in order to maintain shipping channels. From sand deposits in the open sea in the vicinity of Hel peninsula, 387 310 $m^3$ was extracted. In 1997, there was extracted a total of 820 310 $m^3$ of sand for beach nourishment.

In 1998, maintenance dredging in ports amounted to 1 713 999 $m^3$. Aggregate dredged from sand deposits in the open sea in the vicinity of Hel peninsula amounted to 88 872 $m^3$, and there was a total of 1 802 871 $m^3$ of sand extracted for beach nourishment.

**Sweden**

During 1995, one new permit for marine exploitation was granted in Sweden. This was the permission for the Öresund Link Consortium to dredge new stretches for part of the Flint shipping channel between Saltholm Island and the coast of southwestern Sweden in connection with the building of the link between Sweden and Denmark. All the material dredged will be used for construction of two islands south of Saltholm Island on the Danish side of the Sound. The total amount of sediments extracted to date is $2.5 \times 10^6$ $m^3$.

Presently, there is one new application for marine exploitation, from the city of Ystad requesting permission to dredge 500 000 $m^3$ over ten years at the bank of Sandhammaren. All the material dredged will be used for beach nourishment.

**United Kingdom**

Production of marine aggregates in the UK in 1998 fell to $22.9 \times 10^6$ tonnes from $24.8 \times 10^6$ tonnes in 1997. The demand for marine aggregates for construction and export was fairly flat in 1998. The drop in total extraction was accounted for by a further decline in the quantity supplied for beach replenishment.

In 1998, $13.42 \times 10^6$ tonnes were used by the construction industry mainly for concrete; $7.04 \times 10^6$ tonnes went for export primarily to the Netherlands and Belgium, with smaller quantities to France and Germany. Marine-derived aggregates and sand continued to supply about 15 % of the total demand in Great Britain during 1998, with the main areas of use concentrated in the southeast, from the Thames Estuary to Southampton, and in the southwest.

There was no calcareous seaweed extracted from Crown Estate land in 1998, although a limited amount of extraction did take place in the Falmouth Estuary under the ownership of the local Harbour Commissioners in 1995. Very small quantities of marine sand and gravel were extracted from non-Crown land.

The management of marine aggregate reserves is becoming increasingly sophisticated, with good quality gravel reserves being blended with lesser-quality material to supply the specified cargo and no more. Sustainability and environmental prudence are key issues that are being addressed as the aggregate industry moves into the next millennium.

For marine sand and gravel, the demand forecast in England (excluding coastal protection and exports) averages $21 \times 10^6$ tonnes per year for the fifteen-year period 1992 to 2006, inclusive. However, in the past seven years, the actual supply to ports of landing in England averaged $11.6 \times 10^6$ tonnes, i.e., only 60 % of the forecast demand.

**United States of America**

The single commercial mining operation for marine construction aggregates is in New Jersey and is producing a little over one million cubic yards per year. Beach nourishment remains a major use. The largest single beach nourishment project in the world (24 million cubic yards) is under way along the New Jersey coast. The ocean disposal site for dredged sediment from the port of New York was recently all but closed and sand is being used to cap the site. Many ports on the East Coast are considering large dredging projects as they compete to be the first deep-water port in the USA.

### 8.1.2 Review of national seabed resource mapping programmes

**Belgium**

The Ministry of Economic Affairs will sponsor the acquisition of a multibeam system that will be installed on board the Belgian oceanographic vessel to study the two zones where sand extraction is currently taking place. A general mapping of the Belgian continental shelf will be undertaken in due course.

**Canada**

Marine geoscience mapping is the responsibility of the Geological Survey of Canada, with projects on the Atlantic, Pacific, and Arctic coasts. Surveys are conducted in the nearshore area and on the continental shelf and slope. As a result of reductions in staff, projects on seabed resource assessment and regional mapping are limited in northern areas. Mapping programmes are presently focusing on the southern areas of Canada, where societal pressures are the greatest.

Federal and provincial cooperative efforts in preliminary aggregate and placer gold assessment in offshore Atlantic Canada have been completed. On the west coast of Canada, surficial sediment mapping programmes have been less systematic, and little is known concerning the placer and aggregate resource potential.

The collection of multibeam bathymetric data is considered the most important first step in seabed resource mapping. Cooperative survey efforts are in place with the Canadian Hydrographic Service to collect this information. Recently mapped areas of the continental shelf include Browns Bank, German Bank, off Cape Breton Island, north Prince Edward Island, the south coast of Newfoundland, and the inner shelf of southwest Nova Scotia. Additional surveys are planned for 1999–2000.

Cooperative projects are also under way to assess the effects of trawling and clam dredging on seabed habitats. Results of these surveys will be a quantitative assessment of the effects of bottom fishing gear on biodiversity, community complexity, and ecosystem reoccupation.

Other projects regarding seabed habitat characterization are planned for Browns Bank, to define and understand the lucrative scallop habitat. The fishing community has embraced the new sea-floor mapping technologies as essential tools for a sustainable fishery and to maximize their operations for efficient and safe fishing practices.

**Denmark**

Mapping of the seabed is an integral part of the systematic resource mapping programme in Danish waters. The mapping programme continues and is concentrated on the North Sea, the Kattegat, and the Baltic Sea. Maps of surface sediments, Quaternary geology, and sand and gravel resources have been prepared. At present, between 80 % and 90 % of potential resource areas in inner Danish waters have been mapped.

In 1997 and 1998, reconnaissance mapping was carried out at greater water depths in the central part of the Kattegat and in the North Sea. The preliminary results indicate the presence of interesting resources in the deeper parts of the areas studied.

Some of the most important stone reefs in Danish waters have been mapped during 1990–1996. Two reports have been published which include surface sediment maps, gravel and stone concentration maps, and descriptions of the biology in the areas.

**France**

IFREMER is mapping surface sediments in the southern part of the Atlantic coast in the Bayonne area. BRGM (French Geological Survey) is mapping the whole of the French territory. More recently, BRGM completed several studies dealing with marine aggregate prospecting taking into account other marine activities, particularly fishing. Three main areas have been studied (Baie de Seine, Loire River mouth, and Charente Maritime). Another French agency is also editing maps in collaboration with the French universities; nine maps have been published so far.

**Germany**

The Federal Maritime and Hydrographic Agency has finished a project in December 1998; the objective was to map and quantify movable sands along the German North Sea coast between the waterline and 20 m water depth. Results will be published in the near future.

**The Netherlands**

Resource mapping within the Netherlands is the responsibility of the National Geological Survey. The Geological Reconnaissance Map Series consists of, inter alia, a surface geology sheet which includes a main map showing the uppermost 10 cm of the seabed. The Geology and Resource Map Series consists of map sheets with both geological information and resource information. Geochemical distribution graphs of surface sediments are being prepared. The aim is to have reliable information on natural background values and their variation and, thus, on human-induced changes.

As in previous years, various beach nourishment and other extraction schemes have resulted in dedicated site surveys. The annual demand for beach recharge materials remains at about $15 \times 10^6$ $m^3$.

**Norway**

A seabed sediment map covering the area off southwestern Norway has been published. The mapping of seabed sediments will continue, including habitat mapping, especially of deep-water corals (*Lophelia* reefs) and recent sedimentary processes.

**Sweden**

The Geological Survey of Sweden is responsible for the seabed mapping of the Swedish Exclusive Economic Zone (EEZ). To date, about 16 % of the area has been mapped. The results are presented in a Marine Geological Map Series containing a seabed sediment map and a subsidiary map dealing with some of the geological sections based on interpreted seismic and sub-bottom profiler graphs.

In a joint Swedish-Lithuanian project, two new maps of the central Baltic Sea were printed in 1998. All countries around the central Baltic Sea have contributed to the two maps, one showing the seabed sediments and the other the bathymetry. These are the most complete seabed sediment overview maps of this area.

### 8.1.3 Review of approaches to environmental impact assessment (EIA) and related environmental research

**Canada**

One recent example of a controversial aggregate extraction project is the Middle Shoal Channel Improvement Project. This is a channel dredging project at the entrance to Bras d'Or Lakes of Cape Breton Island, Nova Scotia. The project was submitted for an environmental assessment and required several authorizations that included the requirement for the proponent to develop an Environmental Management Plan to ensure that any potentially adverse effects of the project were identified early and action taken to minimize impacts. A key component of that plan includes an Environmental Effects Monitoring and Compliance programme.

The dredging operation began in August 1996 but was suspended two months later by a Federal Court order after the Union of Nova Scotia Indians requested a judicial review of the Canadian Environmental Assessment Act review of the project. Dredging was approximately ninety percent complete at that time. The conclusion was that the monitoring programme does not allow complete discernment of the impact of the dredging operation on the movement of fish in and through the channel. It was not possible to reach consensus on the impact on fish populations that use this body of water. Permits issued for the dredging have since expired and the proponent has not made an application for renewal that might lead to a requirement for further assessment. The concerned company, that uses the channel to transport gypsum from its mine, has suggested that it may close its mining operations in the area.

**Denmark**

Nearly all the dredging operations have been finished for the Øresund Link. Thus far, only minor effects have been demonstrated. A detailed resource assessment and an environmental impact assessment of the dredging of sand fill have been carried out. The assessment has been prepared in accordance with EC Directive 85/337. Preliminary results indicate, in accordance with the EIA, that there is no environmental impact outside 1000 m from the dredging area.

A major enlargement of the harbour of Århus will require the dredging of more than $5 \times 10^6$ $m^3$ of sand. The Harbour Authority was requested to carry out an environmental impact assessment in accordance with the EC Directive as part of the application procedure. Based on the assessments, spill limits were set at 6 % and 7 % at the two areas to be dredged.

Investigations of the potential impact on bottom fauna and birds are being conducted off the Limfjord prior to the commencement of the dredging of sand to be used in the construction industry.

In order to ensure the sustainable extraction of raw materials, the Danish National Forest and Nature Agency has initiated a three-year research project on environmental economics. One of the aims of the project is to establish a decision framework to evaluate the environmental consequences of existing and future dredging projects based on the content of fines in the resource, hydrography, spreading of fines, and ecological models. A detailed study of the ecological consequences of dredging in coarse sediments has been started, especially concerning the effects on the benthic flora and fauna on surrounding stone reefs. Another monitoring programme has been initiated off the west coast of Jutland to study the effects of the dredging of sand for beach protection.

**France**

Data collected for WGEXT over the past several years have been compiled in a review paper presented at the 1998 ICES Annual Science Conference. The paper was entitled 'Physical and biological impact of marine aggregate extraction along the French coast of the eastern English Channel: Short- and long-term post-dredging restoration'.

**Germany**

A three-year research project was initiated in March 1999 that will focus on refilling processes of extraction pits and furrows off the islands of Rügen (Baltic Sea) and Sylt (North Sea) with respect to changes in sediment composition and distribution, seabed morphology, as well as possible lowering of the seabed and effects on coastal stability.

The field investigation phase of a research project in the Baltic Sea has been completed. The results have not yet been fully analysed, but the following preliminary conclusions can be drawn. The analysis is based on a case study monitoring the physical and biological changes in the water column and the sediment at an extraction area close to the Darss-Zingst peninsula in the southern Baltic Sea. The effects of extraction were very severe, but localized, and were found to have a short-lived impact on the macrobenthic populations within the area. In the dredged area, the populations were reduced in abundance, diversity and biomass, but the area was not totally devastated. The following summer these changes led to localized oxygen depletion in the deeper parts of the extraction area and especially in the dredged furrows. To date, a comparable phenomenon has not been described for these flat and dynamic coastal regions. As a consequence of the oxygen depletion, there was a second die-off of the macrobenthic community and a cessation of the recolonization process. Oxic conditions were re-established in the autumn after storm activity, which allowed recolonization of the macro-invertebrate community to continue.

The study also examined the effects of disturbance on several 'sensitive' macrobenthic species found in the red data lists. Preliminary data indicate that none of the three red list invertebrates sampled in the area prior to dredging have recolonized the area.

A literature review is under way on the physical effects of dredging and disposal in marine areas. A preliminary assessment indicates that suspended sediment concentrations and the overburden of sediment on the seabed are among the major factors impacting the benthos. In another study, investigations are planned to measure the effects of turbidity on the behaviour of the common mussel (*Mytilus edulis*).

**The Netherlands**

A beach nourishment project was conducted on the central Dutch coast in 1996/1997. Before and after the nourishment, an ecological monitoring programme of the benthic fauna at the borrow pit was carried out. A survey showed that fifteen months after the refill of the borrow pit with coarser sand from deeper water, the benthic community had for the most part recovered. However, there are still differences in composition and density of the community between the borrow pit and the reference area. Compared to the period before the extraction, the biodiversity at both the pit and the reference area showed reduced values.

The final report of a project on the ecological effects of subaqueous sand extraction north of the Island of Terschelling has been completed. Overall the surveys show that, although the original community structure has not been re-established, the effects of sand extraction that were still visible after two years, are no longer obvious after a period of four years. Natural fluctuations in the benthic community are most likely to explain this departure from the original structure. The changes in the benthic community at the borrow site show concurrence with patterns found in the entire coastal and offshore zone in the Netherlands.

Several plans for large land reclamation projects are being launched in the Netherlands. Studies are being undertaken to define the morphological and ecological effects of large-scale extraction. The effects on the water movement, sand transport and morphology are defined by the location, orientation to the main currents, and the design of the extraction area.

**United Kingdom**

The report 'Regional Seabed Sediment Studies and Assessment of Marine Aggregate Dredging' has been prepared to provide a structured approach to regional studies of sediment mobility over the seabed which can be used in other parts of the UK, identifying the objectives, benefits, scope, and methods that can be used. It also provides guidelines for studies needed to assess the impacts on the coastline of proposed dredging operations.

Another report, entitled 'Seabed Sediment Mobility West of the Isle of Wight', describes an investigation of the sediments and their mobility over a large area of seabed. The results of such a study can help planners assess the possible environmental effects of proposed dredging of the seabed for sand and gravel.

### 8.1.4 Seabed characterization and biotope mapping

To manage the marine environment effectively, maps that reveal the geophysical characteristics of the seabed are essential, since they allow the wide-scale geology and modern-day sedimentary processes to be determined and understood.

The most commonly used, highly developed, and versatile systems are the side-scan sonars and multibeam swath bathymetric devices. Side-scan sonar has been defined as an acoustic imaging device used to provide wide-area, large-scale pictures of the seabed. Multibeam bathymetry is a relatively new seabed mapping technology that can be applied to an understanding of marine habitats, aggregate resources, and seabed processes. Another device, the sub-bottom profiler, provides high-resolution definition of the seabed sediments down to about 50 metres. It also offers the potential to map infaunal communities and to examine the interactions between the benthic fauna and sediments.

A summary of the advantages and disadvantages of the main acoustic systems currently available is provided in Table 8.1.4.1, along with an overview of other seabed mapping tools. WGEXT concluded that for broad-scale mapping of aggregate biotopes (> 1 km$^2$), either 'chirp' based side-scan sonar or multibeam swath bathymetry offered the most cost-effective means of discriminating different sediment types and dynamic processes, given the area which can be covered or surveyed at relatively high resolution. For small-scale biotope classification (< 1 km$^2$), high resolution side-scan sonar (cm), underwater cameras, and grab sampling methods were considered the most appropriate mapping tools.

### 8.1.5 Effects of increased suspended solids caused by dredging

Sediment plumes arising from marine aggregate extraction are perceived to lead to impacts in the marine and coastal environment. A number of research studies have been undertaken or commissioned to assess these impacts.

Research has included studies to assess the amount and characteristics of material released from overspill and from screening, assessment of the behaviour and fate of the plume, and the development of predictive models for sediment dispersion.

The ACME agreed that further consideration should be given to understanding the effects of suspended sediments on fish and benthic invertebrates, and to determining the movement of material close to the seabed. It is also apparent that further research is necessary to refine the existing predictive models.

**Table 8.1.4.1.** Overview of seabed mapping tools.

| System | Coverage | Resolution (horizontal) | | | | | | | | Remarks |
|---|---|---|---|---|---|---|---|---|---|---|
| | $km^2\,hr^{-1}$ | km | 100m | 10m | m | dm | cm | mm | <mm | |
| Remote sensing, satellite | >100 | x | x | x | | | | | | Restricted to satellite operation coverage and to shallow areas (not more than 6 m water depth) |
| Remote sensing, aircraft | > 10 | x | x | x | x | | | | | Only for shallow areas (not more than 6 m water depth) |
| Multi-beam | 3–6 | x | x | x | x | x | | | | Allows the use of backscattering data for analysing bottom substrate |
| Single-beam | 1–2 | x | x | x | x | x | | | | No surface coverage |
| Side-scan | 1–8 | | x | x | x | x | | | | Size of surface coverage depends on the frequency used |
| Sub-bottom profiler | 0.5–1 | x | x | x | x | x | | | | No surface coverage |
| Video camera | 0.1–0.2 | | | | x | x | x | | | Only site inspections, allows epibenthos identification |
| Sediment profile camera | < 0.001 | | | | | | x | x | | Only site inspections |
| X-ray photography | < 0.001 | | | | | | x | x | | Only site inspections, allows more detailed analysis than the profile camera (water content, density, etc.) |
| Macro grab/corer sampling | < 0.003 | | | | | x | x | x | | Quantitative data on the flora and fauna requires additional analysis in a laboratory |
| Mieo grab/corer sampling | < 0.003 | | | | | | | x | x | Quantitative data on the flora and fauna requires additional analysis in a laboratory |

Danish research has also indicated that different dredgers discharge varying amounts of sediment depending on the design of the dredger. Further consideration should be given to this area of research to establish the suitability of using different types of dredgers to extract different sediment types.

### 8.1.6 Effects of marine aggregate extraction on higher trophic levels, in particular, fishes and fisheries

The ACME noted the WGEXT discussion of the results of some work using simple fishery models. This work aimed at establishing the extent to which effects on reproductive processes (r-type effects) rather than environmental carrying capacity (K-type effects) should be of concern in discussions of effects on fisheries. The conclusions from this work were:

1) a reduction in either r or K will result in a reduction in sustainable yield from a fishery;
2) the scale of the consequent reduction in sustainable yield is dependent on the extent of exploitation of the fishery;
3) the greatest reduction in sustainable yield will occur where fishing effort is beyond that required for the maximum sustainable yield (MSY), and for reductions in r rather than K;
4) the effects of natural variability in reproductive capacity on exploited stocks is disproportionately greater at effort levels beyond those required to achieve the MSY, and slight reductions in r, were they to occur as a result, for example, of aggregate extraction disturbance sources, may here also disproportionately affect the risk of stock collapse.

The conclusion is that the focus for concern should be the unregulated fisheries, and regulated fisheries that are considered to be very heavily fished, where aggregate extraction may potentially have effects on reproductive processes rather than on environmental carrying capacity. It should be noted that there is an inherent presumption in this model that effects will negatively impact the fishery; this may not apply in all cases.

**Consideration of effects in terms of the biodiversity of marine fishes**

WGEXT analysed a database of all fishes in the Northeast Atlantic area (excluding the Baltic Sea) with the aim of developing a short list of fishes of greatest concern in relation to marine extraction activities and consequent threats to the biodiversity of marine fishes. Taking first those most at risk from effects on critical habitats (i.e., spawning or nursery areas), WGEXT noted that the species in Table 8.1.6.1 may need further consideration

**Table 8.1.6.1.** Bottom-spawning species of fish occurring within the geographical range of aggregate extraction activities.

| Species name | English common name | Depth (m) Minimum depth range | Maximum depth range | Bottom spawning description |
|---|---|---|---|---|
| *Alosa fallax* | Twaite shad | 0 | 100 | Bottom spawning? in tidal reaches of rivers |
| *Ammodytes marinus* | Raitt's sandeel | 30 | 150 | Presumed to lay eggs on sand and fine gravel |
| *Ammodytes tobianus* | Sandeel | 0 | 30 | Lays eggs in sand which adhere to the sediment |
| *Aphia minuta* | Transparent goby | 0 | 60 | Lays eggs in empty bivalve shells |
| *Apletodon microcephalus* | Small-headed clingfish | 0 | 25 | Demersal eggs on kelp holdfasts |
| *Blennius pavo* | No known common name | 0 | 30 | Lays eggs in crevices and hollowed debris |
| *Blennius rouxi* | Striped blenny | 0 | 30 | Eggs laid under stones guarded by males |
| *Buenia jeffreysii* | Jeffrey's goby | 10 | 330 | Eggs found in mollusc shells—guarded by male |
| *Chromis chromis* | Damselfish | 0 | 40 | Territorial, eggs laid on bed and guarded by male |
| *Clupea harengus* | Herring | 0 | 150 | Oviparous - demersal eggs |
| *Crystallogobius linearis* | Crystal goby | 5 | 400 | Lays eggs on seabed in worm tubes at around 30 m |
| *Diplecogaster bimaculata* | Two-spotted clingfish | 0 | 55 | Demersal eggs on stony grounds |
| *Eleginus navaga* | Navaga | 0 | 20 | Spawns in 8–10 m over rocky or sandy bottoms—eggs sink |
| *Gobius couchi* | Couch's goby | 5 | 5 | Presumed eggs laid in rocky crevices |
| *Gobius cruentatus* | Red-mouth goby | 0 | 5 | Presumed eggs laid on undersides of stones |
| *Gobius gasteveni* | Steven's goby | 36 | 74 | Presumed eggs laid under stones on seabed |
| *Gobius niger* | Black goby | 2 | 70 | Eggs laid on underside of shell debris or stones, etc. |
| *Gymnammodytes semisquamatus* | Smooth sandeel | 20 | 200 | Laying eggs over sand gravel (or coarse sand) |
| *Hyperoplus lanceolatus* | Greater sandeel | 0 | 150 | Laying eggs in sand—larvae pelagic |
| *Lebetus guilleti* | Guillet's goby | 2 | 30 | Presumed to lay eggs on shells or stones |
| *Lebetus scorpioides* | Diminutive goby | 30 | 375 | Presumed eggs laid on shells or stones |
| *Lepadogaster candollei* | Connemarra clingfish | 0 | 5 | Demersal eggs laid on the underside of stones |
| *Lepadogaster lepadogaster* | Shore clingfish | 0 | 25 | Demersal eggs on underside of boulders |
| *Lesuerigobius friesii* | Fries' goby | 20 | 350 | Presumed to lay eggs on seabed debris—muddy ground? |
| *Mallotus villosus* | Capelin | 0 | 100 | Demersal eggs on gravel in shallow coastal waters |
| *Muraena helena* | Moray (eel) | 0 | 200 | Not known |
| *Myoxocephalus scorpius* | Bull-rout/Father lasher | 4 | 60 | Eggs laid on seabed and guarded by males |
| *Myxine glutinosa* | Hagfish | 0 | 150 | Not known |
| *Pomatoschistus lozanoi* | Lozano's goby | 0 | 30 | Eggs deposited on empty bivalve shells—guarded by male |
| *Pomatoschistus microps* | Common goby | 0 | 5 | Lays eggs in hollow of inverted bivalve shells |
| *Pomatoschistus minutus* | Sand goby | 0 | 20 | Lays eggs in empty bivalve shells guarded by the male |
| *Pomatoschistus norvegicus* | Norway goby | 30 | 280 | Presumed to lay eggs in shells or under stones |
| *Pomatoschistus pictus* | Painted goby | 0 | 50 | Lays eggs on bivalve shells guarded by male |
| *Pungitus pungitus* | Ten-spined stickleback | 0 | 5 | Eggs laid in nest made by male and guarded |
| *Raja alba* | White skate | 40 | 370 | Oviparous, demersal eggs |
| *Raja batis* | Common skate | 30 | 600 | Oviparous, demersal eggs |
| *Raja brachyura* | Blonde ray | 0 | 100 | Oviparous, demersal eggs |
| *Raja clavata* | Roker | 5 | 280 | Oviparous, demersal eggs |
| *Raja fullonica* | Shagreen ray | 35 | 500 | Oviparous, demersal eggs |
| *Raja microocellata* | Small-eyed ray | 0 | 100 | Oviparous, demersal eggs |
| *Raja montagui* | Spotted ray | 25 | 120 | Oviparous, demersal eggs |
| *Raja naevus* | Cuckoo ray | 20 | 150 | Oviparous, demersal eggs |
| *Raja undulata* | Undulate ray | 0 | 200 | Oviparous, demersal eggs |
| *Scyliorhinus canicula* | Lesser-spotted dogfish | 3 | 400 | Lays eggs in shallow water |
| *Scyliorhinus stellaris* | Nursehound/Bull huss | 1 | 70 | Lays eggs in shallow water |
| *Serranus hepatus* | Brown Comber | 0 | 100 | Specific site chosen to lay eggs then guarded by male |
| *Spondyliosoma cantharus* | Black sea-bream | 0 | 20 | Eggs laid in nests/hollows on seabed—guarded by male |
| *Thorogobius ephippiatus* | Leopard-spotted goby | 6 | 40 | Presumed eggs laid in crevices |

**Table 8.1.6.2.** Species associated with sand and gravel habitats that may be subject to extraction activities.

| Species name | English common name | Depth (m) | | Seabed habitat description |
|---|---|---|---|---|
| | | Minimum depth range | Maximum depth range | |
| *Alosa fallax* | Twaite shad | 0 | 100 | Pelagic but bottom spawning in gravel |
| *Ammodytes tobianus* | Sandeel | 0 | 30 | Found close to and burrowing in clean sandy seabed |
| *Arnoglossus imperialis* | Imperial scaldfish | 60 | 100 | Mainly on sandy or muddy grounds |
| *Arnoglossus laterna* | Scaldfish | 10 | 60 | Mostly on sandy bottoms |
| *Arnoglossus thori* | Thor's scaldfish | 15 | 92 | Mainly on mud and sandy bottoms |
| *Atherina presbyter* | Sand-smelt | 0 | 20 | Demersal, abundant on mud and sandy bottoms |
| *Blennius pavo* | (Not known) | 0 | 30 | Rocky shores, coasts, in sands and muds |
| *Buglossidium luteum* | Solenette | 5 | 40 | Mostly on sandy bottoms |
| *Callionymus reticulatus* | Reticulated dragonet | 0 | 40 | Living on clean sandy bottoms |
| *Dasyatis pastinaca* | Stingray | 3 | 75 | Bottom living on soft sand, less so on muddy ground |
| *Echiichthys vipera* | Lesser weever | 0 | 50 | Lies buried in sand foraging seabed for food |
| *Gobius cruentatus* | Red-mouth goby | 0 | 5 | Found on stones and sandy ground among eelgrass |
| *Gobius gasteveni* | Steven's goby | 36 | 74 | Found on muddy sand and with shell and small stones |
| *Gobius niger* | Black goby | 2 | 70 | Found on sandy and muddy bottoms and eelgrass |
| *Limanda limanda* | Dab | 20 | 40 | Mostly on sandy grounds migrating inshore to spawn |
| *Mallotus villosus* | Capelin | 0 | 100 | Shallow gravel areas form the spawning grounds |
| *Microchirus variegatus* | Thickback sole | 37 | 92 | Found on sand, and sand gravel bottoms |
| *Mullus surmuletus* | Red mullet | 3 | 90 | Sand and mud and also rocky bottoms, digs for prey |
| *Mustelus asterias* | Starry smooth hound | 0 | 70 | Mainly bottom living on sand and gravel grounds |
| *Myliobatis aquila* | Eagle ray | 0 | 100 | Part bottom living on sand or mud but also surface |
| *Platichthys flesus* | Flounder | 0 | 55 | Mostly on sandy and muddy bottoms |
| *Pomatoschistus lozanoi* | Lozano's goby | 0 | 30 | Mostly on coarse sand and muddy bottoms |
| *Pomatoschistus microps* | Common goby | 0 | 5 | Mostly on muddy sandy bottoms and in rock pools |
| *Pomatoschistus minutus* | Sand goby | 0 | 20 | Mostly on sandy grounds |
| *Pomatoschistus pictus* | Painted goby | 0 | 50 | Mostly on gravel, shell and coarse sand with stones |
| *Raja brachyura* | Blonde ray | 0 | 100 | Bottom living mostly on sandy sediments |
| *Raja microocellata* | Small-eyed ray | 0 | 100 | Bottom living mostly on sandy grounds |
| *Raja radiata* | Starry ray | 50 | 100 | Bottom living on sand, mud and occasionally gravel |
| *Raniceps raninus* | Tadpole-fish | 0 | 100 | Bottom feeding on sand and mud and algae on rock |
| *Scopthalmus maximus* | Turbot | 0 | 80 | On shell gravel, gravel and sand |
| *Scopthalmus rhombus* | Brill | 9 | 73 | Mostly on sand but also gravel and mud |
| *Serranus hepatus* | Brown Comber | 0 | 100 | Among rocks, sand and sea grasses |
| *Serranus scriba* | Painted Comber | 0 | 30 | Among rocks, sand and sea grasses |
| *Solea solea* | Sole/Dover sole | 10 | 100 | Mostly sand and mud but active mid-water nocturnal |
| *Torpedo marmorata* | Marbled electric ray | 10 | 100 | Entirely bottom dwelling on sand and rarely mud |
| *Trachinus draco* | Greater weever | 8 | 100 | Lies buried in sand, forages at night |
| *Tripterygion atlanticus* | Black-face blenny | 0 | 20 | LW and deeper among stones and fine gravel, hides |
| *Zoarces viviparus* | Eelpout/Blenny | 0 | 40 | Rocky shores in pools and deeper on mud and sand |

With regard to those species more generally associated with sand and gravel habitats subject to extraction activities, a separate list was identified for further consideration. Table 8.1.6.2 lists species identified as occurring principally in depths less than 100 m, and in association with seabed habitats of either sand or gravel (a total of 38 species).

It was noted that the above lists should be considered indicative.

**Cumulative environmental impacts of aggregate extraction**

A four-year study in the UK, which commenced in January 1998, is intended to investigate the potential for cumulative effects of multiple dredging activities on the seabed environment and fisheries. Preliminary findings indicate that a combination of bathymetry and sediment particle size are the main factors governing the distribution of biological communities. Future work will include relating the biological patterns to factors such as sediment mobility due to waves and tides.

Finally, the ACME took note of the discussion about herring spawning grounds and requested that WGEXT examine ways in which such areas could be properly identified and protected in accordance with the ICES Code of Practice for the Commercial Extraction of Marine Sediments. It was agreed that this could be taken up with the Herring Assessment Working Group and other appropriate Working Groups and pursued as an action for the next WGEXT meeting.

## 8.2 Data Requirements to Estimate the Environmental Effects of the Disposal of Fish Offal and Discards in the Baltic Sea

*Request*

Item 5 of the 1999 requests from the Helsinki Commission: to provide information on the possible impact of dumping of fish remnants, especially regarding:

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

*Source of the information presented*

ACME deliberations.

*Status/background information*

The ACME considered the requirements of this request in detail and identified the types of data needed to address this request, as follows:

1) an estimate of the total magnitude of fish remnants discharged from sea-going fish-processing units by ICES rectangle, if possible, but at least by ICES Subdivision;
2) the annual amount of discards from fishing vessels, with the appropriate length composition of the discards, by pelagic and demersal species, and by ICES rectangle;
3) the composition and abundance of possible consumers (birds, fish, benthos) of fish remnants and discards by sea area, as well as estimates of consumption rates;
4) estimates of the effect of bacterial decomposition of discards and fish offal in the coastal zone and below the halocline.

*Recommendations*

ICES strongly recommends Member Countries to make their data on discards available to ICES for collation and further analysis.

## 8.3 Progress in Preparing a Chapter on Baltic Fish Stocks, Diseases, and Ecosystem Effects

*Request*

Item 3 of the 1999 requests from the Helsinki Commission: to prepare a chapter on 'Baltic fish stocks, diseases and ecosystem effects' for the Fourth Periodic Assessment of the State of the Marine Environment of the Baltic Sea, 1994–1998 comprising:

a) an update of the material prepared by ICES for the Third Periodic Assessment concerning (i) commercial fish stocks, (ii) coastal fish, and (iii) diseases and parasites of Baltic fish; and
b) information on the effects of mariculture (including the genetic effects of releases of cultured fish and possible use of genetically modified organisms); and
c) information on the ecosystem effects of fishing activities in the Baltic Sea (taken from the material provided by ICES in 1997 in response to a HELCOM request, plus any new material).

*Source of the information presented*

The 1999 reports of the Baltic Fisheries Assessment Working Group (WGBFAS), the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO), the Working Group on Environmental Interactions of Mariculture (WGEIM), and the Working Group on the Application of Genetics in Fisheries and Mariculture (WGAGFM), and ACME deliberations.

*Status/background information*

The ACME reviewed the progress by the above-mentioned Working Groups in the preparation of material for the chapter on Baltic fish stocks, diseases, and ecosystem effects that will be submitted for inclusion in the HELCOM Fourth Periodic Assessment. The ACME noted that in most cases the material under preparation will comprise an updated version of a chapter prepared by ICES for the HELCOM Third Periodic Assessment of the State of the Baltic Marine Environment, that covered the years 1989–1993. However, WGEIM has outlined a plan to collect a considerable amount of new material on mariculture-related issues. In addition, the Working Group on Ecosystem Effects of Fishing Activities (WGECO) will prepare its contribution to this chapter when it meets at the end of November 1999.

According to the schedule requested by HELCOM, the update will be produced intersessionally for review at the meetings of WGBFAS, WGPDMO, WGEIM, and WGAGFM early in 2000. After this review, the material will be compiled by the ICES Secretariat for review of the fisheries portions by the Advisory Committee on Fishery Management (ACFM) in May 2000. The overall chapter will be reviewed by ACME in June 2000.

## 8.4 ICES Environmental Report

*Request*

There is no request. During the past few years, several ICES Working Groups have agreed to contribute to an ICES Environmental Report, which will be updated annually or more frequently, depending on the subject matter. The Environmental Report is published on the ICES website as material becomes available.

### 8.4.1 Oceanographic conditions

*Source of the information presented*

The 1999 report of the Working Group on Oceanic Hydrography (WGOH) and ACME deliberations.

*Status/background information*

An ocean climate monitoring network, consisting of fifty standard stations and sections located at strategic points around the North Atlantic, is maintained by eight ICES Member Countries. The Working Group on Oceanic Hydrography (WGOH) decided during its 1999 meeting to follow up the success from 1998 and continue to prepare a summary of oceanographic conditions in order to provide fishery and environmental managers a brief update, describing the present status of the physical environment within ICES waters. The objective of this year's summary is to present a simplified version of national reports, in a non-technical manner, in order to set the oceanic context of the year 1998. It is intended to be of use to managers of any aspect of the marine environment in the ICES area. In addition to the presentation of data, some expert interpretation has also been added.

Since the North Atlantic Oscillation (NAO) is known to control or modify three of the main parameters which drive the circulation in the North Atlantic (wind speed, air/sea heat exchange, and evaporation/precipitation), a knowledge of its past and present behaviour forms an essential context for the interpretation of observed ocean climate change in 1998. The NAO alternates between a 'high index' pattern, characterized by strong mid-latitude westerly winds, and a 'low index' pattern in which the westerly winds over the Atlantic are weakened. High index years are associated with warming in the southern North Atlantic and Northwestern European shelf seas, and cooling in the Labrador and Nordic Seas. Low index years generally show the reverse. Much of the North Atlantic reflected the 1997/1998 weakly positive winter NAO index by experiencing average or moderate conditions. Areas, which responded to the 1996 reversal of the NAO index, showed a return to high NAO conditions in 1998/1999. There is presently no evidence of a causal connection between the El Niño Southern Oscillation (ENSO) phenomena and the NAO. Both appear to respond quite independently of one another.

In the Northwest Atlantic, air temperatures were warmer than average. There was less winter sea ice, and the duration of ice cover was less than normal. West Greenland experienced warm sea temperatures and a mild climate for the area. Off Newfoundland, the warm period that commenced in 1996, following the NAO reversal, continued into 1998. In general, warmer than normal conditions were experienced in Icelandic waters during 1998. This gave biologically favourable conditions in the waters around Iceland. Also in the Rockall Trough and in the Faroe Bank Channel, high temperatures were observed in 1998. In the Rockall Trough, the upper ocean salinity reached a maximum for the 24-year record, exceeding values observed during the high salinity years of the 1980s. Finally, the North Sea and the southern part of the Norwegian Sea were warmer and more saline than on average. The northern part of the Norwegian Sea and the Barents Sea, however, showed temperatures close to the long-term average. The eastern Barents Sea was even colder than average. However, at the very end of 1998, increased temperatures were observed in the entire Barents Sea.

In the Bay of Biscay, the temperatures were high in the winter of 1997/1998, followed by low temperatures in summer. The summer temperatures are linked to the strong upwelling observed due to prevailing northeasterly winds. The ongoing trend of increasing salinities in the upper layer since 1995 continued.

### 8.4.2 Harmful algal blooms

*Source of the information presented*

The 1999 report of the Working Group on Harmful Algal Bloom Dynamics (WGHABD) and ACME deliberations.

*Status/background information*

The maps produced by WGHABD in 1999 are more complete than those in previous years due to the efforts of ICES Member Countries. These maps are attached as Annex 2. The US maps will be updated next year.

Because of a possible misuse of the information presented and the seafood market sensitivity, WGHABD proposed that a warning be included on each map and that a liability disclaimer be included in the general presentation text, as given below:

GENERAL DISCLAIMER—WARNING

These maps present information on toxin presence from 1989–1998. The information is based on annual national reports by ICES Member Countries. The information available on individual events varies greatly from event to event and from country to country. The monitoring intensity, number of monitoring stations, number of samplings, etc., also vary greatly and, therefore, there is not a direct proportionality between recorded events and actual occurrences of, e.g., toxicity in a given region. Furthermore, areas with numerous recorded occurrences of harmful algal events (HAEs), but with efficient monitoring and management programmes, may have very few problems and a low risk of intoxication, whereas rare HAEs in other areas may cause severe problems and represent significant health risks.

Therefore, these maps should be interpreted with caution with regard to the risk of intoxication by seafood products from the respective areas/regions/countries.

IOC and ICES are not liable for the possible misuse of this information.

The warning to be included on each decadal map of toxin presence states the following:

'DISCLAIMER—WARNING

This map should be interpreted with caution with regard to the risk of intoxication by seafood products from the respective areas/regions/countries. IOC and ICES are not liable for possible misuse of this information.'

The ACME noted that WGHABD has proposed a reporting format for toxic events to be used, after a probationary period, by scientists around the world.

During the coming months, a selected number of scientists from WGHABD have agreed to test the functionality of the HAE database (HAEDAT). After this period, the database will be operational and each Member Country should use it as the main reporting tool. Information will be provided through the Internet at the IOC website in Vigo, Spain.

*Recommendations*

ICES strongly supports the IOC initiative to build a database on harmful algal events. It was noted that the reporting format is adequate, but the functionality of the database should be tested during a probationary period with a view to progressively automating the decadal map production process.

ICES recommends that Member Countries produce their 1999 national reports automatically from the HAE database.

*Additional comments*

ICES ACME expresses its gratitude and appreciation for the high quality work done by the members of the WGHABD on producing the national reports, with a special mention to Ms C. Belin and her assistant B. Raffin for compiling the reports and producing the decadal maps.

# 9 MARINE CONTAMINANTS: DISTRIBUTION, AVAILABILITY, AND EFFECTS

## 9.1 Identification of 'New' Contaminants in the Marine Environment

*Request*

There is no specific request; this is part of the continuing ICES work to keep under review contaminants of interest in a marine environmental context.

*Source of the information presented*

The 1999 report of the Marine Chemistry Working Group (MCWG) and ACME deliberations.

*Status/background information*

ICES has been concerned with the identification of 'new' contaminants in the marine environment for over three decades. MCWG has contributed consistently to the work on identification of 'new' contaminants of concern and, in this connection, considered an overview of 'New contaminants and their relevance to the marine environment', prepared by Prof. B. Jansson (Sweden). This overview summarizes some current activities for assessing risks and setting priorities for regulating chemicals in the OSPAR Commission and other regulatory bodies. The ACME agreed to reproduce this overview, in somewhat amended form, in the paragraphs below, along with a summary of the MCWG discussion based on this overview.

The term 'new contaminants' is often used to describe contaminants that have not been studied extensively in the environment. Those that have been studied extensively are referred to as 'classical' contaminants and include families such as DDTs and PCBs. The discovery of 'new' contaminants is often made by environmental chemists in survey studies, but can also result from new signals appearing in the monitoring of classical contaminants.

Another important way to identify compounds that may be 'new' contaminants is to look at the properties of chemicals produced and used in society. The number of substances, or groups of substances, presently in use is not known but is estimated to be several tens of thousands. The list of compounds in use, as declared by the industry in the European Union in 1981, contains more than 100 000 entries, but it can be anticipated that not all of them are used in significant amounts. To set priorities among this large number of chemicals is a major task and the ranking must be based on political decisions. Some of the priorities already set or presently being discussed are described below.

In the OSPAR Convention of 1992 (see the OSPAR website at http://www.ospar.org for further information), the Contracting Parties agreed to take persistence, toxicity, and bioaccumulation into account in the prioritization process for chemicals. These criteria have led to the identification of heavy metals, organohalogen compounds, organic compounds of phosphorus and silicon, biocides, and oils as substances of concern. Subsequently, OSPAR has, through expert judgements, produced a list of substances of primary concern.

In the Esbjerg Declaration of the 1995 North Sea Ministers Conference, it was underlined that the signatory parties should aim for 'continuously reducing discharges, emissions and losses of hazardous substances to the North Sea and thereby moving towards the target of their cessation within one generation with the ultimate aim of concentrations in the environment near background levels for naturally occurring substances and close to zero concentrations for man-made synthetic substances'. The definition of hazardous substances is not given, but the same parameters as in the OSPAR Convention were mentioned.

The task to identify these hazardous substances was given to OSPAR, which established the Ad Hoc Working Group on the Development of a Dynamic Selection and Prioritisation Mechanism for Hazardous Substances (DYNAMEC). DYNAMEC has started to make a list of compounds that are of possible concern for the marine environment. They will apply a ranking algorithm to the compounds on that list to produce a ranking of relative risk. The properties used for this ranking are persistence, bioaccumulation, and toxicity and different cut-off values will be investigated. After this ranking, a selection of compounds for priority action will be made. In parallel with the ranking procedure, a 'safety net' will provide possibilities to include compounds of special interest, for example, compounds that occur in the marine environment or have endocrine disrupting effects.

Within the UN Economic Commission for Europe Long-Range Transboundary Air Pollution (ECE LRTAP) Convention (see http://www.unece.org for more information) two new protocols were signed in 1998, one for metals and the other for persistent organic pollutants (POPs). These protocols include lists of substances and the measures signatory parties must take to reduce the risk for transboundary transport via air of these compounds. The sixteen substances or groups of substances in the POPs protocol were selected using persistence, bioaccumulation, and toxicity, but also the presence of a compound in remote areas was taken into account.

A global agreement is presently being negotiated within the UNEP POPs programme (see the UNEP website at http://irptc.unep.ch for further information). Initially there are twelve substances or groups of substances on the list for action, but there will be a possibility to add more substances to the list if they fulfill specified criteria. A special 'Criteria group' is presently working to set cut-off values for persistence, bioaccumulation, and toxicity; occurrence in remote areas is also included.

A new directive has been proposed by the European Commission to harmonize legislation regarding water quality within the EU. Water quality is planned to be measured using both biological and chemical parameters. At present, work is in progress to decide which parameters to use, and for the chemicals a prioritization process similar to that in OSPAR will probably be used. Measured data are intended to play an important role in the priority setting.

A great deal of resources are spent on risk assessments of chemicals and a complete evaluation of one substance is very expensive. This means that prioritizing the compounds to be studied is very important. This has so far mainly been done by expert judgements and there are lists of compounds to be evaluated. The risk assessments in the EU follow a strict scheme detailed in a technical guidance document. A modelling tool is used to estimate exposure if there is a lack of measured data. There is a risk that these estimated exposures become very conservative and that resources to take measures against environmental and human risks are not spent in an optimal way. Alarm signals in the environment must play an important role in the setting of priorities among chemicals to be studied. Compounds found in remote areas indicate that they are persistent enough to survive a long transport, often through several environmental media. Another alarm signal is when a certain compound is found at increasing levels over time, as this indicates that it is being emitted faster than it can be degraded and sooner or later may reach effect levels. Adverse effects appearing in the environment that cannot be explained may also be due to chemicals and serve as a serious warning that must be investigated.

Environmental chemists have several important functions in this work with chemicals, including:

1) monitoring whether measures taken against emissions of a chemical have had the desired effect in the environment;
2) identification of new contaminants that appear in the analyses;
3) supporting the priority-setting process;
4) supporting risk assessments by providing exposure data:
   a) making all results produced easily available;
   b) initiating and participating in survey programmes for chemicals to be assessed.

In the discussion of this overview by MCWG, a number of comments were made concerning screening procedures for new contaminants. It was noted that a set of guidelines on how to proceed with this type of work would be useful and that this type of work should be coordinated as much as possible by central organizations such as the marine conventions. The identification of new contaminants is also quite different from monitoring, because for a first approach, an estimate of the concentrations within one order of magnitude can be sufficient. The importance of linking new contaminants with their sources was also emphasized. Frequently, the chemical industry will not reveal its sources, or diffuse sources might be involved. The lack of information from the industry is also one of the problems encountered by the OSPAR Working Group on Inputs to the Marine Environment (INPUT). Apparently, the chemical industry will reveal its production figures for risk assessment purposes, although this information generally remains confidential.

MCWG reviewed the OSPAR 1998 list of candidate substances that was made available for its meeting. It appeared that not all the compounds on the list are toxic or persistent. This emphasizes the importance of prioritization procedures. It was noted that biological effects are an important factor in risk assessment, together with exposure levels. However, the toxicology of many of the compounds in question is not well enough known for a proper evaluation, although routine toxicological tests are generally part of the evaluation processes.

Based on the precautionary principle, the Swedish government has proposed that within ten years products in Sweden should not contain persistent compounds, irrespective of whether they are toxic or not. With regard to the alarm systems mentioned in the overview, it was noted that the presence of a compound far away from its source illustrates the importance of Arctic research. Looking for increasing concentrations may often prove difficult or even impossible; specimen banks would be useful in this respect and should be encouraged by ICES. There are a number of new initiatives currently under way. In Germany, non-target screening for new contaminants is being undertaken. In Japan, another approach is used; when a laboratory develops a method, it is distributed to other laboratories who are encouraged to analyse a limited number of samples. In this way, an extensive list of compounds that can be measured is compiled which should be further investigated. This approach was considered to be generally applicable for other areas also.

In conclusion, the ACME recognized the importance of looking for new substances and recommended that this should be one of the objectives of ICES. A clear distinction should be made between this type of work and routine monitoring. Some scope should be created in monitoring programmes to look for new contaminants.

## 9.2 Overviews of Contaminants in the Marine Environment

*Request*

There is no specific request; this is part of the continuing ICES work to keep under review contaminants of interest in a marine environmental context.

*Source of the information presented*

The 1999 report of the Marine Chemistry Working Group (MCWG), the review notes submitted by MCWG, and ACME deliberations. MCWG coordinates the preparation of overviews on contaminants that may be of interest in a marine environmental context. The papers that pass this review process are transmitted to ACME for further consideration.

*Status/background information*

At its meeting in 1999, MCWG examined review notes on several compounds and classes of compounds for which there is interest in a marine context, in order to evaluate their significance in the marine environment. The following describes the current status of this work.

**Tributyltin**

At its 1999 meeting, MCWG discussed a review note entitled 'The Environmental Distribution and Effects of Tributyltin—an Update to 1998', prepared by R. Law (UK) and E. Evers (The Netherlands). The review note had been prepared with particular emphasis on aspects related to the continued use of tributyltin (TBT) antifouling paints on large sea-going vessels. The note summarizes the current knowledge of the fate, distribution, and effects of tributyltin in open sea areas, shipping lanes, and close to ports and harbours. This review note was accepted by ACME and is attached as Annex 3. A brief summary is given below.

The review note shows that reduced TBT concentrations have been observed in coastal areas and marinas frequented by small craft but not in sites affected by inputs from large vessels. Similarly, although some affected dogwelk populations have recovered since the partial ban on TBT was enacted, effects can still be observed in populations close to harbours receiving large vessels. Recent studies along major shipping routes, however, indicate that water concentrations of TBT in areas subjected to large vessel traffic are still elevated relative to those in the open sea. The cessation in the decline of TBT concentrations in water in many areas, its continued impact on distant, non-target organisms, and the persistence of butyltins in sediments, as well as the associated problems for the disposal of dredged sediments, all provide support for an extension to the partial ban on TBT to include its use in antifouling paints for vessels larger than 25 m in length.

*Recommendations*

In view of the extensive and clear evidence that the continuing use of tributyltin on large vessels is contaminating many marine areas at concentrations above the safe level for molluscs, and in view of the equally clear evidence that molluscs in the vicinity of large vessel activity are continuing to experience adverse effects, ICES recommends that the remaining uses of tributyltin on vessels with a waterline length greater than 25 m should be phased out as soon as practicable. ICES supports the recommendation made by the International Maritime Organization (IMO) Marine Environmental Protection Committee at its 42nd meeting that the application of organotin-based antifouling paints should be prohibited from 1 January 2003 and that they must be phased out altogether by 1 January 2008.

**Synthetic musk compounds**

A paper entitled 'Polycyclic Musk Fragrances in the Aquatic Environment', prepared by Dr G.G. Rimkus (Germany), was forwarded by MCWG. The ACME considered this paper to be interesting, but noted that most data presented concerned freshwater environments. The ACME requested additional information on concentrations of these substances and their effects in the marine environment. A brief summary of this paper is provided below.

Synthetic musks are used worldwide as fragrances in perfumery. The first synthetic musk fragrances, the group of so-called nitro musks, were introduced at the beginning of this century; the use of polycyclic musk fragrances began in the 1950s and their production has increased annually since then. At present, the share of polycyclic musk compounds is about 70 % of the total world market of synthetic musks and their production volume has already reached about 6000 tonnes per year. A decline in nitro musk production is occurring due to the restrictions on their use caused by some toxicologically adverse effects. The polycyclic musks are indane and tetraline derivatives which are highly substituted, mainly by methyl groups. The significantly lipophilic and possibly persistent nature of polycyclic musks, together with their tendency to bioaccumulate in fish and other aquatic organisms, underlines the need for development of knowledge on the concentrations and behaviour of synthetic musks in the marine environment.

Polycyclic musks are used in many consumer products, including perfumes, cosmetics, soaps, shampoos, laundry detergents, and other household products. Most of these substances enter the freshwater environment, primarily via sewage discharges. Concentrations of 1,3,4,6,7,8-hexahydro-4,6,6,7,8,8-hexamethylcyclopenta[*g*]-2-benzo pyrane (HHCB) and 7-acetyl-1,1,3,4,4,6-hexamethyl-tetrahydro-naphthalene (ATHN), the two main substances of polycyclic musk compounds, are present in fresh water up to a mean of 2500 ng $l^{-1}$ and in freshwater biota up to 91 mg $kg^{-1}$ lipid weight. However, although they are bioaccumulative, they are also capable of being rapidly metabolized and depurated. There appear to be few data on their persistence or on their toxicity to aquatic life.

There are very few data on concentrations of polycyclic musks in the marine environment, but information from the German Bight shows that concentrations in sea water are low ($< 5$ ng $l^{-1}$), as are concentrations in molluscs and crustaceans ($< 0.4$ mg $kg^{-1}$ lipid weight). On the basis of the limited toxicity data available, predicted no-effect concentrations (PNEC) for HHCB and AHTN are 6800 ng $l^{-1}$ and 3500 ng $l^{-1}$, respectively. This suggests that the concentrations of polycyclic musks present in the southern North Sea may be substantially below hazardous levels. However, there are insufficient monitoring and toxicity data available to permit a reliable risk assessment for marine ecosystems.

*Need for further research or additional data*

In view of the fact that the risks to marine organisms posed by polycyclic musks cannot be assessed at present, the ACME recommends that research should be conducted in the following areas:

a) collection of data on concentrations of polycyclic musk compounds in marine organisms, sediments, and waters throughout the ICES area. Special attention should be given to estuarine and coastal areas receiving large volumes of sewage and industrial effluents;

b) measurement of the acute and chronic toxicity of polycyclic musks to a variety of marine organisms, which can be used to develop a more reliable risk assessment;

c) measurement of bioconcentration factors of polycyclic musks in aquatic organisms other than fish;

d) measurement of degradation rates of polycyclic musks under marine environmental conditions.

*Recommendations*

ICES recommends that further studies be conducted on the distribution and biological effects of musk compounds in the marine environment.

*Additional comments*

The ACME expresses its appreciation for the review prepared by G.G. Rimkus, Official Food and Veterinary Institute Schleswig-Holstein, Neumünster, Germany.

**Polybrominated biphenyls and polybrominated diphenylethers**

A paper entitled 'Polybrominated biphenyls (PBBs) and polybrominated diphenylethers (PBDEs)', prepared by J. de Boer, K. de Boer, and J.P. Boon (The Netherlands), was reviewed and accepted by ACME. It is attached as Annex 4. This paper presents a review of the current state of knowledge of the two groups of compounds, including their chemical and physical properties, production and use, consumption, emission, transformation, distribution, environmental levels, and toxicology. Particular emphasis is given to the occurrence of PBDEs in sperm whale blubber, which is seen to be indicative of contamination of the deep oceans. These compounds are also widely distributed in other marine mammals, in birds, and in fish. The paper also provides a summary of the uncertainties and gaps in the information on PBBs and PBDEs in the marine environment.

PBBs and PBDEs are used as flame retardants, and their many congeners are generally persistent, lipophilic, and bioaccumulative. Their acute toxicity is fairly low, but PBBs have a variety of long-term effects, e.g., they can promote carcinogenicity, interfere with the endocrine system, and suppress the immune system. Less is known about the long-term toxicity of PBDEs, although they may have effects on the reproductive, endocrine, and immune systems. Almost all these data are derived from studies with rodents, so it is difficult to extrapolate to marine organisms and conduct a reliable environmental risk assessment.

*Need for further research or additional data*

The ACME noted that there is an urgent need for information on the long-term toxicity of brominated flame retardants to marine organisms such as fish, birds, pinnipeds, and cetaceans. Particular attention should be focused on their potential to interfere with the balance of the endocrine system.

*Recommendations*

ICES recommends that the research needs identified above should be urgently addressed by the marine science community.

## 9.3 Influence of Biological Parameters on Concentrations of Trace Metals in Fish Liver

*Request*

There is no specific request; this is part of the continuing ICES work on the monitoring of contaminants and the assessment of results from monitoring programmes.

*Source of the information presented*

The 1999 report of the Marine Chemistry Working Group (MCWG) and ACME deliberations.

*Status/background information*

A study on the relationship between biological parameters (fish length and the size, fat and water content of the liver) and the concentrations of trace elements and/or major elements in the liver of cod was presented in the 1998 ACME report (ICES, 1999). That study was limited to Icelandic cod. Data in the ICES database on the liver of cod from the Baltic Sea, the Kattegat, the Skagerrak, the North Sea, and the Norwegian Sea have now been treated in the same way as the Icelandic cod data to investigate whether the model found for Icelandic cod has a more general applicability. It was found that this was generally the case, although there were some deviating results in the Baltic cod data sets. If the model is found to apply to cod from different regions, then it has a wide application in the evaluation and assessment of monitoring data.

*Need for further research or additional data*

Further work utilizing the data in the ICES data bank is required, with the aim of publishing the results in the open scientific literature. This, however, requires that permission from the countries that have supplied the data to ICES is obtained for their use for this purpose.

**Reference**


ICES. 1999. Report of the ICES Advisory Committee on the Marine Environment, 1998. ICES Cooperative Research Report, 233: 69–70, 275–283.


## 9.4 Processes that Post-Depositionally Enrich or Deplete Metals and Organic Contaminants in Sediments

*Request*

There is no specific request; this is part of the continuing ICES work on review and update of information on sediment chemistry, and physical, biological, and chemical processes that may influence the concentrations of metals and organic contaminants in sediments.

*Source of the information presented*

The 1999 report of the Working Group on Marine Sediments in Relation to Pollution (WGMS) and ACME deliberations.

*Status/background information*

In 1999, WGMS was requested to report on the impact of processes that post-depositionally enrich or deplete metals and organic contaminants in sediments.

Information on the progress being made in the understanding of these processes is considered necessary in environmental studies using marine sediments, e.g., for the interpretation of sediment monitoring data, for distinguishing natural from anthropogenic metals in sediments, for better understanding of the bioavailability of sedimentary contaminants, and for interpretation of retrospective trends of contaminants from the sedimentary record.

Physical, chemical, and biological processes in the vicinity of the sediment-water interface influence the chemical composition of the sediments. Diagenesis refers to the sum total of these processes. In its 1995 report, the ACME reviewed the influence of diagenesis on the distribution of cadmium in marine sediments, and recommended great caution in the interpretation of cadmium data from surface sediment samples analysed as part of monitoring programmes (ICES, 1995).

The nature and content of organic matter and the redox conditions in the sedimentary environment play a major role in the binding of a number of compounds to the sediments and in their exchange with interstitial water and the water column (including organic contaminants, metals, and phosphates). However, in addition to such diagenetic processes influencing the water column, sedimentary chemical conditions are in turn influenced by physical, chemical, and biological processes in the water column.

Valuable information may be obtained from carefully executed monitoring programmes on contaminants in surface sediments with a view to establishing spatial distributions, and identifying contaminant gradients and areas of concern, as well as concentration changes over time. Nevertheless, in all cases, a better knowledge of the early diagenetic processes at the sediment-water interface will contribute to a better understanding of the fate of contaminants in the sedimentary environment and support the interpretation of the results of sediment analyses.

The ACME raised a question about the current level of understanding of these processes in relation to organic contaminants, particularly regarding the influence of redox conditions. It was acknowledged that very little is currently known regarding the diagenesis of organic contaminants in sediments.

The ACME noted that to take into account the additional complexities associated with sediment diagenesis, it is necessary to analyse sediment cores instead of taking surface sediments with grab samplers.

*Need for further research or additional data*

The ACME recognized that, in general, diagenetic processes are not considered often enough as factors influencing the concentrations of trace metals and organic contaminants in marine sediments, and thus more attention should be given to understanding these processes in future research programmes.

**Reference**

ICES. 1995. Report of the ICES Advisory Committee on the Marine Environment, 1995. ICES Cooperative Research Report, 212: 81.

## 9.5 New Methods and Research Regarding Biological Effects of Contaminants

*Request*

There is no specific request; this is part of ongoing ICES work to measure and interpret the effects of contaminants in the marine environment.

*Source of the information presented*

The 1999 report of the Working Group on Biological Effects of Contaminants (WGBEC) and ACME deliberations.

*Status/background information*

The ACME took note of various new methods and research initiatives regarding biological effects of contaminants, as summarized below.

**Direct Toxicity Assessment of waters and discharges**

The use of biomarkers and bioassays for assessing the biological quality of effluents being discharged to marine waters was considered. This growing area is termed Direct Toxicity Assessment (DTA), and is now recognized to be important for screening complex discharges where chemical monitoring alone is unlikely ever to be able to provide sufficient control. DTA methodology was first developed and applied in the USA, but is now being tested in the UK for both freshwater and marine discharges. Several countries including Canada and the USA have DTA experience, but only with freshwater discharges to date. The most important aspect of DTA methods is that they should provide a rapid and sensitive result, but thus far biomarkers have only been used rarely and acute or sub-acute endpoints are still the norm. There are also a number of technical issues to be solved concerning temporal variability of sample quality and salinity adjustment of brackish or freshwater samples to be tested with marine organisms. There have been a number of recent technical innovations, particularly in Norway, in the technology used for deploying fish in cages in effluents, receiving waters, and the open sea, and although these methods are relatively expensive, they give a very useful integrated picture of exposure over sometimes extended periods. Such methods have an advantage over measuring pollution effects in wild fish because the test organisms will only have been exposed to local contamination. In general, DTA techniques applied to marine discharges are likely to be much more widely used in the future, but there is a need for the development of more sensitive and rapid endpoints.

**Research programmes on marine endocrine disruption**

WGBEC reviewed the impacts associated with chemicals of concern in the marine environment, in particular considering endocrine disruptors in UK and Dutch estuarine and coastal waters (the EDMAR and LOES programmes, respectively). The UK EDMAR programme follows up on earlier observations of strongly oestrogenic effects in UK flounder (*Platichthys flesus*) and, as well as investigating the causative substances, is studying possible effects on fish populations. It is also broadening into studies of androgenic effects in fish, and both oestrogenic and androgenic effects in crustaceans. It is too early to know whether the endocrine disruption occurring in some UK marine fish is serious enough to threaten the long-term future of exposed populations. The Dutch LOES programme is a large baseline study to investigate oestrogenic compounds in the aquatic environment. It is investigating the sources and identity of endocrine disrupting substances, as well as measuring impacts on sentinel fish species in estuarine, coastal and offshore areas. A pilot survey was carried out in 1997/1998 which showed that natural hormones and xeno-oestrogens are present in Dutch estuarine and marine waters and are able to exert generally weak oestrogenic effects.

**Molecular biological methods in marine monitoring**

WGBEC revisited the subject of new methods in molecular biology that can be applied to marine monitoring, that was first addressed in 1995. Molecular methods were defined as methods 'that use the detection and/or quantification of single biological molecules through binding of external reagents', and they are generally able to detect the effects of contaminants in very small volumes of tissue or body fluids. Such methods are used with RNA and DNA by employing primers or probes, proteins using antibodies, and any cellular component using specific dye probes and, in the case of bioassays, cell lines transfected with receptor and reporter genes. Since 1995, considerable methodological progress has been made in the following areas:

1) the analysis of immunochemical data (image analysis tools);
2) the use of immunoassays (radioimmunoassay (RIA), enzyme-linked immunosorbent assay (ELISA)) in monitoring (e.g., metallothionein (MT), vitellogenin (VTG));
3) the use and understanding of membrane transporters in fish (multixenobiotic resistance (MXR), multidrug resistance (MDR)).

The major molecular methods that are currently available are listed in Table 9.5.1. The ACME noted the opinion of WGBEC that molecular techniques have a considerable future in marine monitoring, but that they have been slower in being implemented than was originally envisaged.

**Table 9.5.1.** Existing molecular methods that could have potential in marine monitoring.

| Method | Biomarker | Status | Explanation | References |
|---|---|---|---|---|
| Immunoassay (RIA, ELISA, western blot) | CYP1A<br>MT<br>VTG | In use | Quantification, identification (protein) | 9<br>6, 10, 25<br>see Table 5.1.1.1 |
| Immunoassay (RIA, ELISA, western blot) | Zona radiata protein<br>spiggin<br>DNA adducts<br>MDR/MXR<br>*ras*, *myc* oncogenes | Research phase | | 1<br>-<br>see Table 5.1.1.2<br>11<br>23 |
| Immuno-cytochemistry | CYP1A<br>MDR/MXR<br>G6PDH<br>GST-A | In use | Quantification, verification (protein) | 15<br>7, 15–16<br>15<br>15 |
| Northern blot, slot-blot | CYP1A<br>MT<br>VTG<br>MDR/MXR<br>*ras*, *myc* oncogenes | Used for explaining modes of action | Quantification (mRNA) | 4, 9, 17<br>3, 13<br>19<br>8, 22<br>2, 21 |
| Subtractive hybridization | - | Research phase | Identify sequence differences (DNA) | 27 |
| *In situ* hybridization | - | Used for explaining modes of action | Quantification, verification (mRNA) | 5 |
| Liquid hybridization | - | Research phase | Quantification (mRNA) | 28 |
| Competitive PCR | - | Research phase | Quantification (mRNA) | 26 |
| Probes | Lipofuschin<br>intracellular Ca<br>membrane transport | Used for explaining modes of action | Quantification, verification | 20<br>24<br>12, 18 |
| Receptor-binding | Oestradiol<br>androgen<br>cortisol | Used for explaining modes of action | Affinity | 14<br>14<br>14 |

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**Proposal for an ICES sea-going workshop on biological effects techniques for pelagic systems**

WGBEC reviewed and approved detailed plans for a sea-going workshop to study the effects of contaminants on pelagic biota. The concept of this workshop has been developing for several years, but it is now anticipated that it will be held for three weeks in March–May 2001. The main objective is to assess the ability of various methods to detect biological effects of xenobiotic chemicals in pelagic systems, particularly in fish larvae and other zooplankton. In addition, the results from the workshop will be available as a basis to suggest methods for future monitoring of biological effects in pelagic systems. It is intended to focus the workshop on areas affected by discharges from the oil and gas industry.

The ACME agreed that there is reason to believe that a practical workshop which brings together international expertise in this area will facilitate the development of biological effects methods for pelagic ecosystems. Little is known about the impacts of contaminants on pelagic ecosystems. However, before such impacts can be understood, it will be necessary to develop and validate methods for studying them. This workshop aims to contribute towards that goal, using known gradients of pelagic contamination associated with the oil industry.

The ACME therefore recommends that ICES support such an international collaborative workshop on the effects of xenobiotic chemicals in pelagic ecosystems. The workshop should be a joint exercise between ICES, national monitoring organizations, and other interested parties. The ACME noted that organization of the workshop is already reasonably well advanced, and a Steering Group has been established under the convenership of Dr K. Hylland (Norway).

## 10 FISH DISEASE ISSUES

### 10.1 Disease Prevalence in Wild Fish Stocks

*Request*

There is no specific request; this is part of continuing ICES work to update present knowledge on trends in the occurrence of diseases in wild fish stocks.

*Source of the information presented*

The 1999 report of the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) and ACME deliberations.

*Status/background information*

The ACME reviewed the relevant sections of the WGPDMO report providing information on new trends in the occurrence of diseases in wild fish stocks based on national reports from ICES Member Countries.

Special attention was drawn to new trends in the distribution of the following two diseases:

Viral Haemorrhagic Septicaemia (VHS)-like virus appears to be widely distributed in wild marine fishes. The classic VHS virus is a known pathogen in freshwater trout farming, causing heavy mortalities. The VHS-like virus in the marine environment has primarily been described from the Pacific coast of North America, where it has been associated with mortalities in Pacific herring (*Clupea pallasii*) and pilchard (*Sardinops caerulea*) and has been isolated from a series of other fishes. In European waters, the virus has been isolated from various marine fish species during recent years. West and north of Scotland, the virus was mainly found in Norway pout (*Trisopterus esmarkii*), while in Danish waters, the Skagerrak, Kattegat, and the Baltic Sea, herring (*Clupea harengus*) and sprat (*Sprattus sprattus*) were the predominant hosts.

A marked increase in the prevalence of skin ulcers in Baltic cod (*Gadus morhua*) was observed in 1998. It was mainly young cod (1- to 2-years old) in the western part of the Baltic Sea that were affected. Although intensive work has been initiated to clarify the aetiology of the ulcerations, the results are still inconclusive.

*Recommendations*

ICES recommends that:

a) owing to the widespread occurrence of VHS-like virus in various marine fish species in the Baltic Sea, the North Sea, and eastern North Atlantic areas, and the reported associations of this virus with high mortalities in fishes off the west coast of North America, Member Countries should collate information on possible effects of this disease on wild fish populations;

b) due to the marked increase in the prevalence of skin ulcers in cod in the western Baltic Sea, further studies on the aetiology should be conducted by Member Countries.

### 10.2 Evaluation of 1998 Fish Disease Data Assessment

*Request*

There is no specific request; this is part of continuing ICES work to develop methodologies for the analysis and assessment of fish disease data and other types of environmental data in ICES data banks.

*Source of the information presented*

The 1999 report of the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) and ACME deliberations.

*Status/background information*

At its 1998 meeting, the ACME discussed a report prepared by WGPDMO providing the results of a statistical analysis of data held in the ICES Environmental Data Centre on temporal and spatial trends in the prevalence of diseases of dab (*Limanda limanda*) and flounder (*Platichthys flesus*) from the North Sea and Baltic Sea (ICES, 1999). From the results, there was clear evidence of the presence of significant and consistent trends in disease prevalence. However, the direction of the trends was not uniform for all areas (based on ICES statistical rectangles) analysed. Some areas showed markedly distinct patterns which deserve particular attention in the coming years. It was concluded that an identification of possible causes for the observed spatial and temporal trends was not possible at that time. Therefore, WGPDMO emphasized the need for a holistic data analysis involving available environmental, oceanographic, and fisheries data from the ICES data banks.

The ACME endorsed the view of WGPDMO that a more holistic statistical analysis was desirable in order to elucidate possible relationships between environmental factors and fish diseases and that the various ICES data banks could serve as a suitable data pool from which the information required could be extracted. It was, therefore, agreed that in 1999 WGPDMO would provide an overview of suitable environmental, oceanographic, and fisheries data available in ICES data banks and, in addition, conduct a pilot study using a selected subset of ICES data for a statistical analysis, in order to assess the practicality and perspectives of a holistic analysis.

The ACME reviewed the relevant sections of the 1999 WGPDMO report providing the overview on suitable ICES data sets as well as the results of the statistical pilot study, as described below. The relevant details are contained in Annex 5.

An overview was presented for the period beginning in 1981 for three North Sea areas (the German Bight, Dogger Bank, and Firth of Forth) which are characterized by the availability of long-term fish disease data sets for dab. Apart from the fish disease data, information was provided on oceanographic data (water temperature, salinity, oxygen content, nutrients), a variety of organic and inorganic contaminants (in water, sediments, and biota), and fisheries data (dab catch per unit effort (CPUE)) extracted from the ICES Oceanographic Data Centre, the ICES Environmental Data Centre, and the ICES Fishery Databanks. From this overview, there is evidence that, in contrast to the fish disease data, there is a striking lack of other long-term data, in particular with regard to contaminants in sediments and biota. However, certain types of data, e.g., oceanographic and CPUE data, are available for almost the entire period, facilitating a statistical analysis.

For the case study, only data from the German Bight area were selected because of their comparatively good coverage. These data were made available by the ICES Secretariat in aggregated form, according to defined criteria. As a first approach to analyse the data, univariate logistic models were fitted. The results revealed a significant relationship between the disease prevalence and the explanatory parameters considered in nearly half the cases tested, possibly an indication of the multifactorial aetiology of the diseases considered (lymphocystis, epidermal hyperplasia/papilloma, acute/healing skin ulcerations), but also a consequence of correlations among the parameters. In a second approach, multivariate models were fitted for three scenarios (long-term, medium-term, and short-term models), according to the time ranges of available data. Again, a number of the explanatory parameters tested were significantly related to the disease prevalence. However, due to different time ranges analysed and the above-mentioned correlations, they were not in all cases identical to the parameters identified in the univariate approach.

*Need for further research or additional data*

The ACME appreciated the outcome of the data overview and the case study and considered it a valuable contribution to ICES efforts over the past years regarding the establishment of standardized procedures for disease data collection, submission, and archiving, and the development of appropriate statistical methods for a comprehensive data analysis. While the earlier report on statistical analysis of fish disease prevalence data (ICES, 1999) focused on the identification of spatial and temporal trends in the ICES fish disease data, the present study addressed for the first time the role of potentially explanatory environmental factors.

Since the results of the case study revealed some significant relationships between fish diseases and environmental factors, the ACME considered it promising to continue the study and to extend it to larger geographical areas and time windows. However, this would require an extended database and the development of additional statistical methodologies adapted to the specific data requirements.

In order to overcome the apparent lack of non-fish-disease data identified in the overview, the ACME suggested that ICES Member Countries should submit relevant data known to exist in national data banks to the ICES data banks.

Additional statistical methodologies should focus on improved interpolation techniques, the use of disease data for all length classes and both sexes, the consideration of time lags and kinetics of biological responses, and the detection of interaction terms. Furthermore, the precision of results should be assessed by appropriate techniques, e.g., a Markov Chain Monte Carlo simulation.

*Recommendations*

ICES encourages Member Countries to enhance their efforts to submit historic and current data held in national data banks to the ICES Environmental Data Centre, the ICES Oceanographic Data Centre, and the ICES Fishery Data Banks according to established ICES procedures, in order to facilitate a more comprehensive holistic analysis of the interactions between natural and anthropogenic environmental factors and the health status of marine organisms.

**Reference**


ICES. 1999. Report of the ICES Advisory Committee on the Marine Environment, 1998. ICES Cooperative Research Report, 233: 297–327.


## 10.3 Status of the M74 Syndrome in Baltic Salmon and *Ichthyophonus* Infection in Herring

*Request*

There is no specific request; this is part of continuing ICES work to update the present knowledge on causes of the M74 syndrome in Baltic salmon and progress in the understanding of relevant environmental factors, and the status of *Ichthyophonus hoferi* infection in herring.

*Source of the information presented*

The 1999 report of the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) and ACME deliberations.

*Status/background information*

The ACME reviewed the relevant sections of the WGPDMO report providing information on the present knowledge on causes of the M74 syndrome in Baltic salmon (*Salmo salar*) and on the status of *Ichthyophonus hoferi* infection in herring (*Clupea harengus*).

In 1998, Swedish and Finnish data indicated a continuation of the decline in the M74 prevalence in Baltic salmon that was reported in 1997. However, there are strong indications that the prevalence might increase again in 1999 in both Sweden and Finland based on observations of an increased occurrence of broodfish with M74 symptoms (wiggling behaviour, pale eggs and flesh) among the 1998 spawners. M74 was not observed in Estonia in 1998.

The ACME noted that, although intensive research has been going on in order to determine the aetiology behind the M74 syndrome (e.g., the Swedish FiRe Project, Bengtsson *et al.*, 1999), no significant breakthrough in this field occurred during 1998. However, new research projects have been initiated to investigate the possible role of toxic algal blooms in the aetiology of the M74 syndrome.

A Theme Session on the M74 Syndrome and Similar Reproductive Disturbances in Marine Animals is on the programme of the 1999 ICES Annual Science Conference in Stockholm.

In 1990, an epizootic of the fungal parasite *Ichthyophonus hoferi* occurred in herring from the Norwegian Sea, the Barents Sea, the North Sea, the Skagerrak, the Kattegat, and the Baltic Sea. Current findings indicate that the *Ichthyophonus hoferi* infection still persists at low prevalence levels in herring stocks in the Kattegat, the northern North Sea, the Norwegian Sea, and the Barents Sea. However, there are reports of increasing levels of infection in Atlanto-Scandian herring and in some catches of North Sea herring. There is no evidence of significant *Ichthyophonus hoferi* infections in herring from the Baltic Sea, the southern North Sea, Icelandic waters, or in the Northwest Atlantic.

No new information on the prevalence of *Ichthyophonus* off the Pacific coast of the USA was obtained.

*Recommendations*

ICES recommends that Member Countries:

a) continue to monitor salmonid populations for the occurrence of reproductive disorders similar to the M74 syndrome;

b) continue to monitor the prevalence of *Ichthyophonus hoferi* infection in herring as a part of the fish stock assessment work.

**Reference**

Bengtsson, B.-E., Hill, C., Bergmann, Å., Brandt, I., Johansson, N., Magnhagen, C., Södergren, A., and Thulin, J. 1999. Reproductive disturbances in Baltic fish: a synopsis of the FiRe project. Ambio, 28 (1): 1–7.

### 10.4 Other Disease-Related Issues

*Request*

There is no specific request; this is part of ongoing ICES work to consider issues related to the aetiology, spread, and impacts of diseases in marine organisms.

*Source of the information presented*

The 1999 report of the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) and ACME deliberations.

*Status/background information*

The ACME noted that WGPDMO had considered a number of other relevant disease issues, three of which are briefly summarized below.

#### 10.4.1 Potential impact of marine biotoxins produced by dinoflagellates and algae on fish populations

Due to the increasing number of reports of mass mortalities in wild and farmed fish populations in many ICES areas that are associated with toxic algal blooms, WGPDMO considered it appropriate to review available information in the light of a possible impact on fish populations from a fish health point of view.

In some fish kills the role of algal toxins is apparent, for example, in connection with mortalities at fish farms. In most cases, however, it has been difficult to positively establish the role of algal toxins due to the limited availability of toxin detection methods and because it is almost impossible to quantify the mortality of wild fish populations due to toxic algal blooms. Losses of larval fish are even less apparent in wild fish populations than those of juvenile and adult fish, but may be quite significant in evaluating population effects of toxic algal blooms. Although deleterious effects have been observed in several fish species when exposed to sublethal concentrations of microalgal toxins under experimental conditions, there is little information on sublethal or subclinical effects in wild fish.

*Need for further research or additional data*

The ACME endorsed the view of WGPDMO that much more basic research is needed before the effects of blooms and algal toxins on wild fish populations can be fully understood. Attention must be especially focused on the following areas of research:

1) the dynamics of microalgal blooms and factors stimulating non-toxic algae to become toxic;

2) the spectrum of bioactive compounds produced by microalgae. Recent research has shown that microalgae produce, in addition to the classical toxins, yet unidentified compounds that are pharmacologically active on fish and fish eggs;

3) the role of microalgal toxins on fish eggs and fry; laboratory tests and field observations have shown that biotoxins can exert pronounced deleterious effects on fish eggs;

4) the subacute and subclinical effects of microalgal toxins on fish; there is a lack of basic knowledge on the potential effects of sublethal exposure, for example, on the immune system of fish;

5) improved diagnostics. Recently developed methods (DNA probes, immuno-histochemical assays) provide possibilities to identify algal toxins in fish tissue and to establish the role of these toxins in fish kills.

### 10.4.2 Disease risk for wild and cultured crustaceans from known pathogens of penaeids

With the increasing economic importance of crustacean farming (mainly of penaeids), there is a growing concern about the introduction and spread of diseases, a problem well known from fish farming for many years.

The greatest threat to the shrimp farming industry, particularly in the USA, has been introductions of exotic viruses into culture facilities via infected broodstock or seed imports. Other sources of pathogens include shrimp feed processed from solid wastes from plants with infected shrimps, the transfer of pathogens from waste in shrimp processing dumps by birds, and the transmission of pathogens by water insects.

While the impact of pathogens in culture facilities is well known, there is little information on the potential impact of pathogens, exotic or enzootic, in wild shrimp populations. It has been shown that wild shrimp species are host to a wide array of infectious agents, although reports of major losses due to pathogens are rare. Wild stocks may be exposed to pathogens occurring in aquaculture through various pathways, including pond effluents, escapes of infected shrimp, losses during transport to shrimp processing facilities, disposal of pond sediment or solid waste, or through movements of infected bait shrimp. Shrimp processing plants pose a potentially serious problem, as shrimp infected with virus have been identified in retail stores in the USA. Over 50 % of the shrimp processed in the USA is imported from Asia where viral diseases are a major problem, suggesting that processing may be a significant means of introduction of exotic pathogens to coastal waters.

*Need for further research or additional data*

The ACME endorsed the conclusions of WGPDMO that:

1) more baseline information is needed on viruses present in wild stocks of penaeid and non-penaeid species;

2) further evaluation is needed concerning the transfer of infective material from shrimp processing plant activities;

3) further work is required on the role of birds and insects as vectors of viruses;

4) the development of more rapid and accurate detection methods for viral infections is needed;

5) the significance of disease outbreaks in aquaculture and the possibility for predicting the potential of those diseases in wild populations need to be examined;

6) shrimp population models are inadequate to explain the observed variability of wild shrimp populations and should be reworked to include a disease factor.

### 10.4.3 Use of parasites of marine fish species as indicator organisms for environmental change

WGPDMO reviewed relevant current information and emphasized that the use of fish parasites as bio-indicators of pollution or environmental change has a potential for application in monitoring programmes. Changes and trends in the occurrence of parasites can be important alarm bell indicators of possible environmental problems. Such information may not determine cause-effect relationships, but may serve to focus the resources of other specialist groups (e.g., toxicologists). However, it was pointed out that, for a more conclusive assessment, further research is required to obtain a better understanding of the complex interactions between parasites, hosts, and the environment. A change in parasite indices can be multifactorial, which causes problems in the interpretation of actual environmental effects. It was further stressed that it is necessary to have a clear objective in using certain parasites as bio-indicators. Some parasites may be highly contaminant-specific, while others will reflect exposure to a wide range of contaminants or other environmental factors. For the detection of trends in the occurrence of suitable parasite species, long-term data sets must be collected, preferably within the framework of already existing fish disease monitoring programmes.

# 11 INTRODUCTIONS AND TRANSFERS OF MARINE ORGANISMS

## 11.1 Status of Ongoing Introductions and Accidental Transfers of Marine Organisms

*Request*

ICES Member Countries may request ICES to review proposed introductions and transfers of marine organisms for mariculture purposes. These proposals receive in-depth review by the Working Group on Introductions and Transfers of Marine Organisms (WGITMO), with final review by the ACME. WGITMO also keeps under review the progress of such introductions and reports the outcome to the ACME.

No new requests for review of proposed introductions were received in 1999, but the status of on-going and proposed introductions and transfers was reviewed.

*Source of the information presented*

The 1999 report of the Working Group on Introductions and Transfers of Marine Organisms (WGITMO) and ACME deliberations.

*Status/background information*

Deliberate releases and accidental introductions of non-native marine species continue to occur in ICES Member Countries. The deliberate releases are carried out for aquaculture or stock enhancement purposes.

**Introductions for mariculture purposes**

The ACME noted the outcome of a four-year study on the effects of the introduction of the red alga *Porphyra yezoensis* in coastal waters off the State of Maine, USA. The conclusions of this study were:

1) *P. yezoensis* plants were present but uncommon in the summer on the shoreline adjacent to the cultivation site during the farming season;
2) local *Porphyra* species out-recruit *P. yezoensis* on netting substrates; and
3) there is no evidence that *P. yezoensis* will over-winter in Cobscook Bay and replace local *Porphyra* species.

It was noted that an assessment is being made of farming a tetraploid (as opposed to the current diploid). This could have ecological implications if there is a request for this new variant to be released to the natural environment for farming purposes. The ACME agreed that the genetic impact of any possible release must be carefully evaluated.

**Accidental introductions and transfers**

The ACME noted that accidental introductions and transfers of non-native species continue to be documented in many ICES Member Countries. These include the invasions described below.

Chinese Mitten Crab, *Eriocheir sinensis*

The Chinese mitten crab, *Eriocheir sinensis*, is gaining new and expanded footholds in European rivers and estuaries, and may become a serious problem. Populations in the Thames River (England) and in German rivers are now exceeding the historical heights that occurred in the 1930s. The Chinese mitten crab is also well established in San Francisco Bay, California, USA, where it is continuing to expand its range.

Veined Rapa Whelk, *Rapana venosa*, in the USA

The veined rapa whelk, *Rapana venosa*, was first identified as present in the Hampton Roads region of the lower Chesapeake Bay on the east coast of the USA in the summer of 1998. The species is native to the Sea of Japan, but was introduced into the Black Sea in the 1940s, and has since spread to the Aegean and Adriatic Seas. There is evidence that this recent range extension is mediated by the transport of early life history stages in ballast water. The current knowledge of the distribution of *R. venosa* in Chesapeake Bay suggests that the majority of the population is limited to a swath from the James River Bridge, through Hampton Roads and along the shoreline inshore of the Thimble Shoals Channel. No specimens have been collected in Maryland waters.

Egg cases of *R. venosa* have been collected from Hampton Roads, and larval forms cultured in the laboratory. Estimates of the salinity tolerance of the early larval stages of *R. venosa* can be used as a precursor to estimating a potential range of distribution of the species within the Chesapeake Bay and its sub-estuaries. These estimates can then be used to examine possible range extensions within the Mid-Atlantic region if a stable, reproducing population becomes established in the lower Chesapeake Bay.

The presence of *R. venosa* in the lower Chesapeake Bay has ecological consequences beyond the obvious potential for predation on commercially valuable shellfish prey species (e.g., the oyster *Crassostrea virginica*, and the clam (quahog) *Mercenaria mercenaria*). In the Black Sea and in their native Sea of Japan, *R. venosa* have been reported primarily from hard-bottom habitats. Adult Chesapeake Bay *R. venosa* have been collected from both hard- and soft-bottom habitats. Laboratory observations indicate that adult *R. venosa* burrow almost completely into the sand at water temperatures > 20 °C (i.e., not over-wintering behaviour). Burrowing behaviour by these large apex predators expands the potential suite of vulnerable prey organisms to include infaunal shellfish (e.g., the bivalves *Mya arenaria, Ensis directus, Cyrtopleura costata*). The presence of large (> 100 mm) empty *R. venosa* shells in Chesapeake Bay may enhance growth of the local hermit crab (*Clibanarius vittatus*). Recent collections of *C. vittatus* from the Hampton Roads area indicate that they use empty *R. venosa* shells as shelters and are reaching unusually large sizes. This has implications of creating abnormally large crustacean scavengers on Chesapeake Bay benthic epifauna (e.g., oyster spat).

*Rapana venosa* has also been reported from French waters and there has been an unsubstantiated report of the gastropod being found in UK waters.

Parasitic Worm *Pseudobacciger harengulae* in Herring

A new parasitic worm, *Pseudobacciger harengulae*, was detected in about 20 % of juvenile herring from the Swedish west coast. Previously this species has been mainly recorded from clupeid fish in tropical to warm-temperate waters of the Atlantic, Pacific, and Indian Oceans, and the Black Sea.

Invasive Bryozoan *Tricellaria inopinata* in UK Waters

A population of the invasive bryozoan *Tricellaria inopinata* has recently (August 1998) been discovered on the central southern coast of England, representing the first Atlantic record for this species. The only other known European record for *T. inopinata* is in the northern Adriatic Sea, its type locality. It is now believed that *T. inopinata* is of Pacific origin, where it is known under two synonyms, at least one of which is shared with another distinctly different morphospecies. Recent systematic reappraisals of the *T. inopinata* 'complex' in the Pacific have revealed its widespread presence in both northern and southern temperate regions, including Japan, North America, Australia, and New Zealand (the last as an invasive species).

Significant differences are evident between the invasion ecology of this taxon in the northern Adriatic and the Northeast Atlantic. The invasion of the northern Adriatic Sea has been slow. After seventeen years, this taxon is still largely confined to the Venice Lagoon, and although it has recently been found in a second lagoon, it has not been recorded on the open coast to date. In the Northeast Atlantic, evidence suggests a recent arrival and a rapid population build-up. It is already abundant within harbours and marinas located in estuarine basins along an 80-km section of the central southern coast of England. A more detailed survey of the Poole Harbour area has revealed its presence in natural habitats, both within the harbour and on the open coast, as well as within marinas. The different ranges of substrates colonized by *T. inopinata* in the Adriatic and Northeast Atlantic are indicative of an opportunistic and flexible substratum selection strategy.

Zebra Mussel, *Dreissena polymorpha*, in Ireland

The zebra mussels probably arrived in Ireland in 1994 or earlier, with the movement of second-hand boats on trailers from either Britain or the Netherlands, or from both countries. Zebra mussels have expanded their range rapidly within the navigable areas of the Boyle, Shannon and Erne systems, a distance of approximately 250 km. Significant increases in abundance have taken place in most of the major lakes in the navigable areas of the afore-mentioned rivers. High densities have been found in the largest lake on the Shannon River, Lough Derg. Here the native unionid clams *Anodonta anatina* are densely covered in zebra mussels, many having >1000 zebra mussels per individual. Large numbers of dead clams, *Anodonta*, have been found in shallow water.

The zebra mussels are readily spread throughout inland navigable waters while attached to a wide range of craft from barges to paddle boats. More than 20 % of the barges sampled were infested. Barges are capable of carrying some millions of zebra mussels and, if they spawn in a region where zebra mussels do not occur, they can produce larvae that can settle to form a new population.

Based on the experience from the range expansions of zebra mussels in Europe and North America, it is likely that these mussels will colonize all navigable waterways in Ireland. However, lake boats, normally used by anglers and frequently carried overland, are likely to bring mussels to other lakes, either attached to boats or on the weeds snagged by trailers. Zebra mussels can survive for several days under favourable conditions. Individuals have survived out of water in March for eighteen days. The transfer to lakes is therefore a very likely mode of introduction.

Water Flea *Cercopagis pengoi* in the Baltic Sea

A recent newcomer to the Baltic Sea, the cladoceran *Cercopagis pengoi*, was first found in the Gulf of Riga in 1992. The species is native to the Ponto-Caspian region and was probably introduced into the Baltic Sea by ships' ballast water. The animal has now spread over the Gulf of Riga and the Gulf of Finland. It occurs very abundantly in calm weather conditions in summer, above a water temperature of 16–18 °C. In these conditions, it can reach a density of 1000 individuals per $m^3$. However, some specimens have been found as late as October at water temperatures of 8–12 °C. The waterflea is actively consumed by several pelagic and bentho-pelagic species in this area, including commercially exploited herring. It is too large to be preyed upon by 0-group fish. Other possible impacts of the species on the ecosystem could include changes in the zooplankton community as a result of direct predation effects and competition for the same food resources with 0-group fishes.

Japanese Seaweed *Undaria pinnatifida* in France

Two recent publications describe the presence of the Japanese kelp, *Undaria pinnatifida*, in the rocky shore ecosystem of the St. Malo area in France. The alga spread quickly in the area after being introduced for farming in the surroundings of the Rance estuary in 1984. The population then extended over several hectares (15 km along the shoreline) below the lower zone of the native brown alga *Fucus serratus*, reaching depths down to 12 m in 1992. After that it retreated, but in 1996 it increased again and also extended its geographical distribution to Normandy in the north.

**National reports on accidental introductions**

Further information was noted on new introductions or range extensions of non-native species, as taken from the national reports of ICES Member Countries. This is summarized below.

Canada

A number of new introductions or range extensions have occurred. The water flea (cladoceran) *Cercopagis pengoi* was discovered along the north side of Lake Ontario in the summer of 1998 and, by late August, these exotic crustaceans were reported throughout Lake Ontario. The amphipod *Echinogammarus ischnus* became common throughout Lake Erie in 1997 and was collected in the St. Lawrence River west of Montreal in 1998. The green crab *Carcinus maenas* was first reported along the eastern end of Prince Edward Island (Northumberland Strait) in the early fall of 1998. It was found by eel fishermen and appears to be at high densities in certain bays. The clubbed tunicate *Styela clava* was first identified on a market mussel crop from the Brudenell River (eastern end), Prince Edward Island, in January 1998.

France

In France a new snail, the oyster drill *Ocinebrellus inornatus*, was observed in 1997 for the first time along the French Atlantic coastline.

Germany

Co-introductions of the seaweeds *Sargassum muticum* and *Ascophyllum nodosum* and the seasquirts (ascidians) *Styela clava* and *Aplidium nordmanni* have occurred in the Wadden Sea with introductions of Pacific oyster, *Crassostrea gigas*, seed from Ireland for mariculture.

There was further spread of the zebra mussel *Dreissena polymorpha*, particularly in the Elbe River.

High concentrations of the Chinese mitten crab, *Eriocheir sinensis*, have been found in German rivers. For example, 3000 kg of juvenile crabs were estimated as the daily catch in the Elbe River. This exceeds the maximum density recorded in the 1930s. The crabs are causing structural damage to riverbanks owing to their burrowing.

Norway

As of 1 January 1999, Norway is part of the European Environment Agency (EEA) agreement, and serves as a controlling body for imports from non-EU countries. Regulations of imports and exports have generally been harmonized with the European Union requirements.

The population of the introduced red king crab, *Paralithodes camtschatica*, in the northern Norwegian coastal area east of the Varangerfjord has grown considerably.

A novel species was reported for Norwegian waters, a red alga *Dasysiphonia* sp. (Dasyaceae, Rhodophyta). It was probably transported accidentally from the North Pacific Ocean (via ballast water or hull fouling) and was observed in the Netherlands in 1994.

Sweden

A new law on environmental issues took effect on 1 January 1999 which will promote sustainable development and protection of nature. It also addresses protection of biodiversity and valuable cultural environments. One chapter regulates the use of Genetically Modified Organisms (GMOs).

In April–May 1998, an algal bloom of a species thus far referred to as *Chattonella* sp. cf. *C. verruculosa* (a species previously known from Japan) occurred in the Skagerrak and northern Kattegat waters and adjacent parts of the North Sea, with reports of fish kills from the Swedish west coast, the Norwegian south and southwest coasts, and the Danish coasts. If the species identification is correct, this is the first record of this species in Europe.

*Recommendations*

ICES recommends that a 'Rapana Alert' be issued immediately to Member Countries, based on the arrival of this voracious large (15+ cm) snail (*Rapana venosa*), capable of consuming commercial clams and oysters, in Chesapeake Bay, USA, a major port system that exports large amounts of ballast water to western and northern Europe. This invasive species, native to the Sea of Japan and subsequently introduced in ballast water to the Black Sea, the Aegean and Adriatic Seas, and most recently to Chesapeake Bay, and with recent records in France as well, may potentially have critical impacts on a global scale.

## 11.2 Ballast Water and Sediment Issues

*Request*

There is no specific request; this issue is of continuing interest to ICES Member Countries and several other international organizations.

*Source of the information presented*

The 1999 reports of the ICES/IOC/IMO Study Group on Ballast Water and Sediments (SGBWS) and the Working Group on Introductions and Transfers of Marine Organisms (WGITMO), and ACME deliberations.

*Status/background information*

With rapidly increasing global trade, and extensive coastal environmental changes, an enormous number of invasions of non-native estuarine and marine plants, invertebrates, and fish have characterized the last quarter of the Twentieth Century. These range from the red-tide causing dinoflagellates to zebra mussels and comb jellyfish, and hundreds of other species. In turn, it is clear that there is an ever-increasing potential for the continued spread of non-native species by shipping activities. The concern for this potential is further reflected in a growing menu of legislative actions aimed at ballast water management, including internationally applicable legally-binding provisions that are now under consideration by the International Maritime Organization (IMO) with a view toward their adoption in the year 2000.

The ACME took note of the key conclusions arising from the third meeting of the ICES/IOC/IMO Study Group on Ballast Water and Sediments, as follows:

1) Ballast water and sediment management must be viewed as a complex multi-layered system. An understanding of the wide variety of vessel types (including ship age and condition, and ballast systems), ballasting history (and thus sources and ages of water and sediments), voyage routes, sea conditions, and durations, the changing biological, chemical, and physical conditions that occur inside ballast systems during transport that may impact biotic survival, and the changing environmental nature of both donor and receiver ports, forms an intricate foundation for ballast research and management.

2) In turn, the choice of sampling devices and instruments (the sampling 'tool box') to assess the biotic diversity of ballast water is closely linked to the nature of the specific research questions being asked and the overall research design. Strong funding support for systematics and taxonomy is absolutely essential for understanding and appreciating the biodiversity of ballast biota. Also critical to assessing ballast biological diversity is continued research on rapid and accurate identification techniques, such as those that may employ molecular probes.

3) Management techniques, including risk assessment science, can and should take a wide variety of approaches and forms, including working closely with ship operators to facilitate an understanding of the problems of exotic species, coordinated with an extensive menu of ballast management guidelines. These guidelines should include port-of-origin management (such as avoiding ballasting during red tides or disease outbreaks), at-sea management (mid-ocean exchange as long as safety permits), and if necessary port-of-arrival management.

4) Due to the ecological, economic, and environmental impacts of ballast-mediated invasions in coastal, estuarine, and inland waters of many countries, ballast water research programmes are increasing in number and depth around the world. It is crucial that researchers (a) collaborate in their research efforts to aim for a cumulative, global, and unified database, (b) establish standardized sampling methods and techniques that ensure the generation of comparative data, and (c) remain in contact with each other through fluent and frequent communication. Facilitating these needs will be international consortia such as SGBWS, continuously updated websites, a joint international ballast website providing links to all research and management groups, cooperative programmes, and training programmes for developing countries to establish and implement effective internationally applicable ballast management provisions.

5) Public, and hence political, awareness and outreach are key components of moving forward with international ballast water management. In this regard, establishing the actual or potential relationships between human health concerns (such as the spread of human diseases, including cholera), economic concerns (such as the spread of shellfish and finfish diseases, the spread of algae causing harmful algal blooms, the spread of species impacting industrial systems, and the spread of species impacting resource utilization and value), and the movement of ballast water and sediments, will remain central to ballast management and science. Included in this is the interpolation of how global changes in climate and the changing coastal and inland water environments globally will either enhance or depress the potential for species invasions.

6) It is recognized that effective ballast management will involve a variety of techniques, approaches, treatments, and controls, played out on a wide variety of spatial and temporal scales. There is urgent need to continue and expand cooperative research programmes that explore the application of management approaches and treatments relative to different vessel types, trade routes, ballast load configurations, weather conditions, and the many other factors that are involved in reducing the transfer

and movement of species. Of particular importance will be the implementation of on-board treatment systems, as they become available and accepted as effective to supplement open-ocean exchange. In all ballast management strategies, it is critical to emphasize the global nature of ballast movement, while acknowledging regional challenges as well.

7) Equally important is the recognition that a ship is a floating 'biological island', and as such is capable of carrying a vast array of living organisms (including bacteria, fungi, protists, animals, and plants) by a variety of means and methods, including (a) as an external surface fouling on the hull, (b) in compartments where water enters the vessel (such as seawater pipes and sea chests), (c) on anchors and anchor chains, and (d) in ballast tanks and ballastable cargo holds. In these 'ship habitats', organisms may be attached to hard substrates, occur in organic films, be suspended in water, or occur in accumulated sediments. An expanded view of the ship as a biological dispersal vector is to recognize both ballast and non-ballast components.

The ACME was informed that an EU Concerted Action started in 1999 on 'Testing monitoring systems for risk assessment of harmful introductions by ships to European waters'. The objectives include to establish a state-of-the-art of European ballast water studies, documentation and intercalibration of ship sampling techniques, an assessment of potential treatment options to reduce the risks arising from ballast water releases, and a public awareness campaign.

In ocean-going workshops of the EU Concerted Action, two vessels were accompanied on their European voyages: the Russian hydrographic ship 'Sibiryakov' was joined on its voyage from St. Petersburg to Lisbon and the oil carrier 'Nordic Torinita' on its trip from Cork to Sture, Norway. During both trips, various sampling methods were applied to several ballast water tanks. The initial results and recommendations are that:

- The temperature variation during short-term voyages, when mostly undertaken in one climate zone, is not a critical factor controlling the survival of the specimens in the ballast water.
- The main factors causing the mortality of species in ballast water during the first days are damage during the pumping process, lack of light (orientation problems), and wind-induced currents in the ballast tank.
- For future ocean-going workshops, the number of sampling techniques should be limited to the most effective methods. During the trips, one sampling trial took about two hours to complete. A limited number of sampling techniques would enable the sampling of one ballast tank more than once a day.
- The cone-shaped net and bucket samples revealed the largest numbers of organisms in both wing tanks that were investigated.
- The recommended methods for zooplankton sampling are the 55 µm cone net, and 55 µm net, bucket and hand pump.
- A new method of using traps inside ballast tanks to catch species in ballast water was not successful. It is possible that a new trap design and a longer period of exposure during better weather conditions will provide better results. The traps should be tested during longer-term ocean-going voyages. Chemical light sticks in traps may attract organisms to the traps.

*Recommendations*

ICES values its cooperation with IOC and IMO in a common study group and recommends the continuation of this cooperation in order to ensure the broadest possible means of handling issues related to the transfer of organisms via ballast water and sediments.

# 12 MARINE BIOLOGICAL PROCESSES AND RESPONSES

## 12.1 Effects of Anthropogenic Nutrient Inputs on the Phytoplankton Community

*Request*

There is no specific request; however, there is an ongoing interest on the part of ICES Member Countries, as well as OSPAR and HELCOM, in information on the effects of anthropogenic nutrient inputs on the phytoplankton community.

*Source of the information presented*

The 1999 report of the Working Group on Phytoplankton Ecology (WGPE) and ACME deliberations.

*Status/background information*

In 1998, the ACME reviewed material prepared by WGPE on the effects of anthropogenic nutrient inputs on the phytoplankton community and asked WGPE to reconsider and revise its material. This request for revision was based on the need for more complete information and evaluation on several aspects, including:

- information on the effective increase in terrestrial nutrient inputs;
- discrimination between different coastal hydrographic systems which imply large variations in the relative importance of terrestrial nutrient fluxes (e.g., the upwelling systems must be considered) and, therefore, of their effects;
- consideration of the possible biases when extrapolating from mesocosms to the real world;
- discrimination between the harmful biomass events and the harmful toxic events.

The effects of anthropogenic nutrient inputs into selected ICES regions were summarized in the 1998 report of WGPE. The common primary feature of nutrification in these regions was the stimulation of increased biomass and primary production; other responses include enhanced growth of green algae (more characteristic of enclosed bays) and shifts in phytoplankton species composition. Secondary effects may also arise due to the decay of organic matter leading to anoxia and mortality of benthos and fish in bottom waters.

This primary stimulation follows yield-nutrient dose principles. Coastal areas are transient zones where fresh, phosphorus-limited water mixes with sea, (mostly) nitrogen-limited water. Consequently, phytoplankton limitation due to nitrogen, phosphorus, or silica has been observed in European coastal waters depending on the season, dynamics of the water body, sediment-water exchanges, N/P ratio in inputs, salinity, etc., and no general rule can be provided. It is not possible to distinguish the threshold of elevated nutrients below which nutrification is a positive stimulus on ecosystem functioning, or above which the nutrient environment becomes degraded leading to negative ecosystem consequences. There is evidence that the intensity of some harmful algal blooms is associated in some areas with eutrophication, but a quantitative relationship has yet to be established, and the mechanisms and bloom species selection have yet to be identified, i.e., whether changes in nutrient ratios or bulk nutrient levels, for example, are responsible. Efforts to apply nutrient resources competition theory to the eutrophication-phytoplankton linkage have been compromised by the difficulty in sorting out the rate constants of remineralization and nutrient delivery rates via riverine systems from chronic build-up effects and the consequences of nutrient storage in different trophic compartments.

There is a general correlation between reduced biotic diversity and phytoplankton communities and eutrophication. Often associated with such biotic shifts are blooms of unusual species or those which become dominant (e.g., *Phaeocystis pouchetii*) in the annual successional cycles.

Watershed management practices, agricultural use of fertilizer, and domestic and industrial waste discharges contribute to the input of nutrients delivered into coastal systems. Efforts to quantify the exact levels of riverine nutrient delivery into the ICES regions lie beyond the scope of the WGPE. Hydrologists, chemists, and experts on coastal processes are more appropriately engaged in collating, analysing, and establishing the nutrient budgets for representative rivers discharging into the coastal waters of the ICES region. However, this effort would not necessarily lead to improved quantification of the linkage between nutrient enrichment and the detected biomass, primary production, bloom species selection and biodiversity owing to the complexity of the phytoplankton controlling mechanisms.

*Need for further research or additional data*

In view of the complexity of the issues and the lack of consensus on the effects of anthropogenic nutrient inputs in coastal waters, the ACME endorsed the WGPE proposal to hold a workshop to address the critical issues. Given the importance of this item in terms of coastal planning, the ACME recommended that a workshop devoted to the effects of anthropogenic nutrient inputs on the phytoplankton community be organized by ICES. It should have a balanced participation of river hydrologists, phytoplankton ecologists, physical oceanographers, ecological modellers, specialists on toxic algae, and phytoplankton monitoring coordinators covering the ICES area.

In addition, long-term data series on plankton and hydrochemistry are needed to investigate this issue in the ICES area, especially to differentiate between the effects of climatic variation and human impact.

*Recommendations*

ICES encourages Member Countries to emphasize the continuation or establishment of long-term data series on plankton and hydrochemistry in the ICES area.

### 12.2 Progress in Understanding the Dynamics of Harmful Algal Blooms

*Request*

There is no specific 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 phytoplankton blooms.

*Source of the information presented*

The 1999 report of the ICES/IOC Working Group on Harmful Algal Bloom Dynamics (WGHABD) and ACME deliberations.

*Status/background information*

The ACME took note of several issues from the WGHABD report, as summarized below.

WGHABD discussed the form, objectives, and functioning of the group in light of the forthcoming IOC-SCOR international programme on harmful blooms under the name of GEOHAB, which stands for Global Ecology and Oceanography of Harmful Algal Blooms. It could be considered that, since WGHABD had sufficiently advanced this theme so that a project like GEOHAB could be launched, its work is done. However, the composition of WGHABD, the geographical area addressed, and the ICES structure could efficiently assist in the development of GEOHAB, especially because WGHABD is considered by IOC and SCOR to have been the mainspring for the new initiative.

WGHABD is special in that it brings biologists and physicists together. However, the need for a larger contingency of physicists and/or modellers is obvious. Fulfilling that need would help in the consideration of recent model developments as well as of a broader range of models than those based on combined biological and hydrodynamic formalisms.

Monitoring is an important aspect of the harmful algal bloom (HAB) problem and WGHABD needs to have an influence on the monitoring programme design and the data interpretation. For example, more environmental data are often needed, and the sampling should be conducted in accordance with local hydrography, taking into account such parameters as mixed layer depth, circulation patterns, frontal dynamics, etc. Historical data and time series from sediments and from climate studies should be examined to look for historical occurrences of HABs (e.g., cyanobacteria in the Baltic Sea for the past 8000 years). It was noted that some HAB problems have decreased in recent times. WGHABD needs to consider decreases in population size (or even disappearances) as well as increases in occurrences or numbers of harmful algae.

The importance of the WGHABD approach and its focus on the population dynamics of the specific species of interest, rather than on phytoplankton ecology in general (which too often implies only the fluxes approach), was emphasized. In fact, flux studies are now finding it necessary to track species-specific differences in order to incorporate their different contributions to flux rates. WGHABD has provided distinct leadership on the problems of HABs and it would be unfortunate to lose the continuity after the larger international programme begins.

The ACME agreed with the WGHABD conclusion that it should continue to focus on population dynamics of the specific species and that it should create more efficient ways of collaborating with physicists.

The ACME noted that WGHABD had elaborated a questionnaire to review the population scenarios for the different harmful algal species. A first analysis revealed that this exercise is more valuable in identifying gaps in knowledge than in helping to elaborate a typology of events. However, this exercise will be highly valuable when designing the implementation plan of GEOHAB.

WGHABD considered the role of harmful benthic microalgae and noted that relatively little attention has been directed to the study of the population dynamics of harmful benthic microalgae. Nor do there appear to be any monitoring programmes for known or potentially harmful benthic species, despite the fact that certain phycotoxin syndromes are clearly linked to benthic/epiphytic species, e.g., ciguatera fish poisoning (CFP). WGHABD reviewed new information, on the basis of a Canadian example, that demonstrates that benthic species can contribute to diarrhetic shellfish poisoning (DSP) contamination. Further case studies in a number of diverse environments are required to confirm whether DSP toxicity caused by benthic microalgae is a widespread phenomenon or is limited to special circumstances. This would have important consequences on monitoring procedures.

As most algae responsible for HABs belong to the dinoflagellates, a group known to produce both temporary and long-term resting cysts, WGHABD identified some critical processes related to cyst-forming species that deserve specific studies. These studies are needed to evaluate the contribution of resting cysts to the development and decline of HABs. It will be necessary to develop coupled biological-physical models for the different HAB species. These models will aid in estimating the importance of location, transport, timing, and size of initial inocula on the development of HABs.

The discussion about modelling was limited to one presentation. The model for the Baltic Sea, from the Kattegat through the Arkona Basin to the Baltic Proper, developed by the Institute of Baltic Sea Research (IOW) in Warnemünde, Germany, reproduces correctly the succession of diatoms, cyanobacteria, and dinoflagellates without using the usual limitation of diatoms by silica. The question of model formulation with or without silica limitation could be dealt with by WGPE. WGHABD considered that such a model could describe the likelihood of a flagellate bloom, but would probably not forecast a specific harmful species.

The ICES/IOC Workshop on the Intercomparison of the *In Situ* Growth Rate Measurements (Dinoflagellates) was held at Kristineberg Marine Research Station in Sweden in September 1996. However, after examination of the available results, WGHABD could only conclude that the results were not comparable and that a synthesis was not possible. A shortcoming of the Workshop was that some techniques had not yet been established and were used for the first time. Rather than proposing publication of the results as an *ICES Cooperative Research Report*, participants were encouraged to publish their findings either in groups or separately.

### 12.3 Zooplankton Ecology Issues

*Request*

There is no specific request; this is part of continuing ICES work on zooplankton issues.

*Source of the information presented*

The 1999 report of the Working Group on Zooplankton Ecology (WGZE) and ACME deliberations.

*Status/background information*

The ACME noted that work has been conducted for a number of years to produce an ICES Zooplankton Methodology Manual. The editorial work has now been completed and the Manual will be published by Academic Press in late 1999.

The Manual consists of a number of chapters authored by selected experts. These chapters present overviews and evaluations of methods within the major fields of zooplankton research. They also give guidance to harmonization or standardization of methods.

The ACME welcomed the completion and publication of the ICES Zooplankton Methodology Manual. This represents a valuable contribution by ICES to the improvement and standardization of methodology for marine research and monitoring.

### 12.4 Benthos Issues

*Request*

There is no specific request; this is part of the continuing work of ICES on benthos issues.

*Source of the information presented*

The 1999 report of the Benthos Ecology Working Group (BEWG) and ACME deliberations.

*Status/background information*

The ACME noted that a Theme Session on Recovery and Protection of Marine Habitats and Ecosystems from Natural and Anthropogenic Impacts had been held at the 1998 ICES Annual Science Conference (ASC). A large range of scientific specialists attended the Theme Session, at which twelve papers and three posters were presented. It was highlighted that:

1) active encouragement should be given to the publication of recovery-type studies, even if they do not conform to recognized models;
2) there is a need to encourage greater understanding of the effect of natural events and not only anthropogenic impacts; and
3) there is a need for better definitions of terminology, e.g., the use and misuse of 'recovery', or the mixed use of the terms 'sensitivity' and 'recoverability'. For non-commercial species and communities, 'importance' and 'significance' require a degree of value judgement to assess, in management terms, 'how much change is acceptable?'

A three-dimensional matrix within which to undertake assessments was suggested. The axes are:

1) sensitivity, recoverability, and importance;
2) type of impact or event;
3) ecological entities (from populations to habitats).

However, assessment is frequently more complicated than the three-dimensional matrix suggested. Expert groups should determine the fields of information required and the criteria or scoring systems to be used. Publications should take advantage of new information technology, especially as relevant texts could become quickly dated. There are now major opportunities to use the Internet to disseminate draft material to relevant experts or make it entirely open to anyone using the World Wide Web. Specialists can then check entries and advise on errors, fill gaps, etc. This was planned in a project beginning in the UK under the Marine Life Information Network (MarLIN) programme.

It was agreed that computer-aided ('expert') systems were useful aids to an experienced marine ecologist in evaluating the likely effects of an activity; however, they must not be seen as a short-cut for use by persons without the necessary expertise.

It was considered important to take account of sublethal effects in assessing sensitivity, for example, effects on reproduction. The effects of TBT on dogwhelks served as a useful example—the dogwhelks developed imposex and could not reproduce, but rather died of old age without producing replacement individuals.

In practical terms, it was felt that recovery is sought to a system that is similar, but not necessarily identical, to the pre-impact habitats and communities. It was particularly important that the functionality of the system (for example, as a nursery) recovered. However, for some species of key importance from the point of view of functioning or for the conservation of biodiversity, the system could not be considered 'recovered' unless those species had returned.

Scientists need to establish quantifiable criteria, even if somewhat pragmatic ones, to assess likely sensitivity and importance (for example, criteria for 'rarity'). Such measures and terms assist politicians and managers who are not scientists to use approaches that are scientifically based without having to understand the scientific details.

At the same ASC, a Theme Session was held on the Evaluation of Marine Protected Areas as Management Tools. The Theme Session focused in particular on a critical evaluation of the effectiveness of area closures in achieving fishery management objectives. While focusing on the effect of Marine Protected Areas on exploited fish and shellfish populations, methodology was presented that could be beneficially applied to other components of ecosystems.

In all of the cases presented, area closures had been established to protect individual targeted fish or shellfish species at particular life stages in discrete geographical locations. The objectives behind the area closures ranged from the protection of juveniles from discarding by mobile gear fisheries, to an overall reduction of fishing mortality and the protection of spawning stock biomass. These studies showed that fishing effort was displaced from the areas closed to fishing to other geographical areas. Some of the expected benefits of area closures were demonstrated in these cases. Generally, biomass increased inside the protected areas following their establishment. However, reductions in total mortality were difficult to detect, particularly at the juvenile stages. While local increases in biomass have been evident in the cases reviewed, increases in stock biomass at the scale of the management unit have not been observed. Benefits achieved within closed areas may be offset by intense fisheries operating in surrounding areas. In cases of seasonal closures, temporally displaced fishing effort may negate the benefits achieved by the stock during its residence inside the closed area. Large-scale environmental effects can also limit the ability to detect the benefits of area closures. In addition, changes in the behaviour of the protected species may occur in response to spatial gradients in exploitation and/or to changes in the productivity of their food resource.

It was concluded that closed areas should not be seen as a panacea for solving fishery management problems. Rather, it appears that overall reductions in fishing effort are required for most stocks. Closed areas should be considered as one of several possible tools to be used in achieving fisheries management objectives and should be tailored to the unique ecological characteristics of the target species. An approach designed to make a quantitative evaluation of closed areas is required, both in relation to temporal changes in the closed area and in relation to control areas. The mixed outcome of the use of closed or protected areas as fishery management tools should not be seen as discouraging the evaluation of the use of these areas in the wider context of ecosystem management.

Finally, the ACME noted that an ICES Symposium on Marine Benthos Dynamics: Environmental and Fisheries Impacts, was held on Crete from 5–7 October 1998. The Symposium attracted about 180 participants from 23 countries. The Symposium centered around three main themes:

1) the evaluation of direct and indirect effects of fisheries-related activities;
2) disturbances due to anthropogenic activities and the recovery of benthos; and
3) natural and disturbance-induced fluctuations in benthic communities.

About forty manuscripts have been submitted for the Symposium proceedings that will be published as a volume of the *ICES Journal of Marine Science* in 2000.

# 13 SEABIRD ISSUES

## 13.1 Seabird Ecology

*Request*

There is no specific request; this is part of ongoing ICES work on the examination of the role and interactions of seabirds in marine ecosystems.

*Source of the information presented*

The report 'Diets of seabirds and consequences of changes in food supply' (ICES, 1999), the 1999 report of the Working Group on Seabird Ecology (WGSE), and ACME deliberations.

*Status/background information*

The ACME noted the growing interest in issues such as food consumption by seabirds in relation to commercial fisheries for forage species, e.g., sandeel, as well as the use of seabirds and seabird eggs in the monitoring of marine contamination (oil, plastic particles, chemical contaminants). Much information and data on these topics have been produced within ICES over the past three years, particularly by the Working Group on Seabird Ecology (WGSE). This information is briefly summarized here.

### 13.1.1 Seabird diet and food supply

The report 'Diets of seabirds and consequences of changes in food supply' (ICES, 1999) is a compilation of major items from the 1997 and 1998 WGSE reports. As the title implies, the focus of the report is on seabird diet and food supply; the report is organized into seven sections which are briefly summarized below.

The first section reviews various issues on seabird food and on the ecological role of seabirds and shorebirds. This section is structured around a number of issues—some issues are dealt with in more expansive detail than others, while some issues identify a series of questions that might be reviewed in the future. In several cases, the recommended reviews will require input from non-ornithological disciplines.

The second section reviews the consumption of pre-recruit fish by seabirds and the possibilities of using such consumption as an indicator of fish stock recruitment. Several good examples exist on the variations in parameters of seabird biology that reflect the importance of pre-recruit fish on seabird diet. This importance is usually indicative of the absolute availability of pre-recruit fish, although sometimes this is only the relative availability. Some seabirds can have a very diverse diet. The answer to the question 'Can seabirds act as indicators of the status of pre-recruit fish stocks?' is 'in some cases'.

In the third section, variation in seabird diet is examined. This analysis is based on a meta-analysis of a database of 767 seabird dietary studies in the ICES area. Issues summarized include annual, seasonal, and spatial variations. Prey selection can differ between closely related species, within a species, and between ages of the same species. Shortcomings in dietary studies were also identified—most particularly, the general lack of large-scale studies offshore and during the winter season.

The fourth section reviews the role of discards from fisheries in supporting seabird populations in the North Sea. There have been two large projects in the North Sea in recent years that have examined discard consumption by seabirds. If seabirds could manage to consume all discards, then some 5.9 million birds would be supported by this food source. This assumption is not correct, but nevertheless, the number of birds supported must be high. The direct effect on scavenging bird species has been an increase in population size, however discard food may not be optimal for rearing young, and the growth in the populations of some birds that also depredate others has probably adversely affected the prey populations.

The possible effects on seabirds of a reduction in the amounts discarded from fisheries are covered in the fifth section. Briefly, the loss in feeding opportunities is expected to place more pressure on other food sources, to change bird population distribution, and to increase competition for food. It is likely that reproductive output would decrease and mortality would increase. In the medium term, population sizes would drop and overall community composition would change.

The sixth section examines the evidence for decadal-scale variations in seabird ecology and links to the North Atlantic Oscillation (NAO). This fairly brief examination of the evidence, using relatively crude tools, uncovered no correlation between the breeding success of seabirds and the NAO index, but several correlations were found between breeding numbers of birds in the Wadden Sea and the winter NAO index. Some interesting areas that further explore these relationships still remain, with such exploration benefiting from interdisciplinary work.

Seabirds periodically suffer mass mortality events that inevitably attract much media attention and doomsday-style headlines. A literature survey was undertaken to determine whether any common underlying features could be discerned in mass mortalities. More than 100 incidents in European waters were identified and the results are detailed in the seventh and final section of the report. These mortality events (or wrecks) are divided into four types: pollution-related (usually oil), weather-related, food-related, and other types. Wrecks were most common in autumn and winter and the most frequently affected species were seaducks and auks. Storms affected the smaller seabirds more than the larger seabird species.

### 13.1.2 Estimation of food consumption by seabirds in the ICES area

At its meeting in 1994, WGSE constructed a model of food consumption by seabirds (not including seaducks or waders) in the North Sea (Tasker and Furness, 1996). This model used information on seabird densities in sections of the North Sea, along with calculated energy requirements, and available information on diet. The outputs of this model indicated that two species, common guillemot (*Uria aalge*) and northern fulmar (*Fulmarus glacialis*), were together responsible for more than 50 % of the total seabird energy requirements. The energy demand was not homogeneous in time or space. Sandeels (*Ammodytes* sp.), fish offal, and discards met about two thirds of the seabird food requirements.

At its 1999 meeting, WGSE conducted further modelling to estimate food consumption by seabirds from other ICES areas, including the Gulf of St. Lawrence, the Northwest Atlantic, Icelandic waters, the Barents Sea, and the Norwegian Sea. Annex 6 provides full details on this preliminary analysis. Major results were as follows:

1) For the Gulf of St. Lawrence, fish accounts for 93.4 % of the annual total prey consumption by seabirds, calculated at 108 419 tonnes. Capelin (*Mallotus villosus*) (36.7 %) and sandeel (22.9 %) represent nearly 60 % of the total consumption, especially by larid and alcid species. Mackerel (*Scomber scombrus*) account for 17.5 % of the prey harvest, but are only taken by northern gannet (*Sula bassanus*). For capelin, the commercial fishery removes only a small proportion of the total biomass.
2) The Northwest Atlantic, in contrast to the Gulf of St. Lawrence, supports large numbers of non-breeding migrant seabirds, probably outnumbering the breeding species. Annual prey consumption is calculated at 707 284 tonnes of fish and invertebrates by breeders and migrants. For breeders, capelin is the most important prey, representing 63.3 %, of which 68.7 % is taken by common guillemot. The mass of capelin taken by seabirds is small compared to the consumption by the main predatory fish and marine mammals, such as cod (*Gadus morhua*) and harp seals (*Phoca groenlandica*). The harvest by fisheries is very small. Sandeel is the second most important species, at 5.1 % of total consumption by breeding seabirds, while various invertebrate species represent 18 %.
3) For waters around Iceland, the annual total prey biomass consumed by the breeding seabird population is calculated at 986 196 tonnes of fish and invertebrates. Northern fulmar and Atlantic puffin (*Fratercula arctica*) are the main consumers, while sandeel, capelin, and euphausiids constitute the major prey species.
4) For the entire Barents Sea, the total annual food consumption by seabirds is estimated at 1 400 000 tonnes. There are large differences among regions in this huge area. The Brunnich's guillemot (*Uria lomvia*) is the major consumer, taking 63 % of the food biomass consumed by seabirds.
5) For the Norwegian Sea, the most numerous species and major consumer of fish is the Atlantic puffin, and Atlantic herring (*Clupea harengus*) is the major prey.

*Need for further research or additional data*

The ACME took note of seabird issues likely to be of importance in the future and, thus, potentially the subject for new research or for the collection of additional data, as identified in the first section of ICES (1999). Topics covered include:

- seabirds as indicators of prey stocks of fish (and shellfish);
- processes affecting the trophic ecology of seabirds;
- seabird and wader interactions with mariculture;
- seabird impacts on the recruitment of fish stocks;
- sources of mortality of seabirds and their relative impact;
- discards and offal as seabird food and potential impacts of changes in discard practices.

**References**

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### 13.2 Seabirds and Marine Contamination

*Request*

There is no specific request; this is part of the continuing work of ICES on monitoring contaminants in the marine environment and assessing their effects.

*Source of the information presented*

The 1999 reports of the Working Group on Seabird Ecology (WGSE) and the Marine Chemistry Working Group (MCWG), and ACME deliberations.

*Status/background information*

In recent years, the ACME has included information on the use of seabirds in monitoring marine contamination, and more specifically seabird eggs for the spatial monitoring of contaminants, in its report (ICES, 1994, 1995). In 1999, two ICES Working Groups (WGSE and MCWG) have produced detailed information about these topics, which the ACME considers very informative for ICES Member Countries and environmental regulatory commissions.

### 13.2.1 Seabirds as monitors of marine contamination

The ACME considered a general review by WGSE on the use of seabirds as monitors of marine contamination and agreed to include it in this report as Annex 7. This review covers the following subjects:

- substances which can be monitored using seabirds;
- advantages of seabirds as biomonitors of contamination;
- drawbacks to using seabirds as biomonitors of contamination;
- criteria for selecting seabird biomonitors;
- recommendations for monitoring contaminants using seabirds.

This information can be used to foster research projects as well as to assist in the development of monitoring programmes for the marine environment.

### 13.2.2 Contaminant concentrations in seabird eggs as environmental indicators

The ACME took note of results of seabird egg monitoring from Sweden and especially from Germany, as considered by MCWG.

In Sweden, the seabird egg monitoring programme has shown that seabird eggs are an appropriate matrix, particularly for the assessment of temporal trends of mercury or organochlorines, but not for some metals such as cadmium. Careful selection of the best species is, however, required due to the migratory nature of some species. Black guillemot (*Cepphus grylle*) eggs are used in Sweden as well as in the Arctic Monitoring and Assessment Programme (AMAP).

In Germany, herring gull (*Larus argentatus*), oystercatcher (*Haematopus ostralegus*), and common tern (*Sterna hirundo*) eggs are used for monitoring. Several research projects have been carried out since 1975. The results of a study on contaminant levels in seabird eggs along the German North Sea coast (Becker, 1989) are given in Table 13.2.2.1.

**Table 13.2.2.1.** Results of monitoring residues of organochlorines and mercury in seabird eggs along the German North Sea coast.

| Topic | Species | Contaminants (mg kg$^{-1}$ wet weight) | Results |
|---|---|---|---|
| Geographical variation (13 sampling sites along the German North Sea coast) | oystercatcher | $\gamma$-HCH (0.002–0.016)<br>$\Sigma$ DDT (0.1–0.5)<br>HCB (0.01–0.26)<br>$\Sigma$ PCB (4–16)<br>Hg (0.2–0.8) | ▪ significant spatial variability<br>▪ higher contamination of samples from the river Elbe estuaries in comparison to the western and northern parts of the North Sea coast |
| Temporal trends (1981–1987) | oystercatcher<br>common tern | $\gamma$-HCH (0.002–0.016)<br>$\Sigma$ DDT (0.1–0.5)<br>HCB (0.01–0.26)<br>$\Sigma$ PCB (4–16)<br>Hg (0.4–2.3) | ▪ significant temporal trends<br>▪ steep rise between 1986 and 1987<br>▪ increase for most of the contaminants investigated between 1981 and 1987 |
| Interspecific differences | oystercatcher<br>common tern<br>Sandwich tern<br>shelduck<br>ringed plover | $\gamma$-HCH, $\Sigma$ DDT, HCB, $\Sigma$ PCB, Hg | ▪ significant differences between species<br>▪ highest levels of almost all contaminants in the fish-eating species (common tern and Sandwich tern)<br>▪ highest $\gamma$-HCH concentrations in shelduck and ringed plover |

Investigations were also carried out in 1996 and 1997 on contaminant residues in bird eggs from the Wadden Sea coast. Residues of PCBs, DDT and its metabolites, hexachlorobenzene (HCB), hexachlorocyclohexanes (HCHs), and mercury were analysed in common tern and oystercatcher eggs. Depending on species, breeding site and year, the results indicate varying concentrations: common tern eggs were generally more contaminated than oystercatcher eggs, which is explained by differences in feeding, breeding, and migration patterns. As monitoring has been carried out in Germany since 1981, long-term trends can be seen: levels for mercury and most organochlorines are decreasing, while lindane ($\gamma$-HCH) levels are either stable or increasing at the oystercatcher breeding sites.

As an important part of German ecological assessment programmes, the Environmental Specimen Bank (ESB) of Germany collects, characterizes and archives a variety of species from representative areas of Germany (Paulus *et al.*, 1996). Among others, the main objectives of the ESB are:

- continuous control of contaminant concentrations in the samples before archiving;
- retrospective analysis of substances which have not yet been recognized as hazardous;
- evaluation of trends in local, regional or global development of contamination.

For an assessment of the contamination of German marine and coastal ecosystems and for retrospective investigations on their development, two representative areas have been selected: the German Wadden Sea National Park (North Sea) and the National Park of the Vorpommern Bodden Area (Baltic Sea). To achieve the above-mentioned goals, a variety of species and matrices have been chosen to characterize the environmental state of these areas, among them herring gull eggs.

According to the standard operating procedures of the ESB, 75 eggs per herring gull colony are sampled annually in the month of May, and some biometric parameters must be determined immediately (length, diameter, weight). In a clean room laboratory, the contents of all eggs from one colony are homogenized and subsamples of 10 g are prepared. These subsamples are stored frozen at −150 °C for archiving purposes. A basic characterization of the herring gull egg homogenate is carried out by determining a variety of organic contaminants and inorganic constituents in some of the subsamples (ESB, 1996), as follows:

- elements and inorganic species (Ba, Ca, Fe, K, Mg, Na, P, S, Sr, Zn, Cd, Co, Cr, Cu, Ni, Pb, Sb, Te, Bi, Sn, Se, As, Tl, Hg, Me-Hg);
- chlorinated hydrocarbons (aldrin, dieldrin, $\alpha$-HCH, $\beta$-HCH, $\gamma$-HCH, HCB, *p,p′*-DDT, *o,p′*-DDT, *p,p′*-DDD, *p,p′*-DDE, OCS, CB28, CB52, CB101, CB138, CB153, CB180, PCP);
- PAHs (anthanthrene, benzo[*a*]anthracene, benzo[*a*]pyrene, benzo[*b*]fluoranthene, benzo[*b*]naphtho[2,1-*d*]thiophene, benzo[*c*]phenanthrene, benzo[*e*]pyrene, benzo[*ghi*]fluoranthene, benzo[*ghi*]perylene, benzo[*j*]fluoranthene, benzo[*k*]fluoranthene, chrysene, coronene, dibenz[*a,h*]anthracene, fluoranthene, indeno[1,2,3-*cd*]pyrene, perylene, pyrene, triphenylene).

Collection, archiving, and basic chemical characterization of herring gull egg homogenates from two sampling sites on the North Sea coast (Elbe estuary and Weser estuary) have been carried out since 1988. Herring gull egg homogenates representative for a Baltic Sea marine ecosystem (one sampling site) have been stored in the ESB since 1991. Table 13.2.2.2 contains examples of temporal trends of selected trace elements determined in herring gull eggs (ESB, 1998).

**Table 13.2.2.2.** Temporal trend monitoring of marine ecosystems carried out by the Environmental Specimen Bank of Germany. Examples of contaminant concentrations in herring gull egg homogenates from two sampling sites at the North Sea coast between 1988 and 1997.

| Parameter | Sampling Site | Concentration (mg $kg^{-1}$ dry weight) | Trend |
|---|---|---|---|
| As | Elbe estuary | 0.35–0.65 | Decreasing at both sampling sites |
| | Weser estuary | 0.2–0.5 | |
| Hg | Elbe estuary | 1.5–4.5 | Decreasing at both sampling sites |
| | Weser estuary | 0.5–1.2 | |
| Cu | Elbe estuary | 2.0–2.5 | Relatively constant, slightly increasing at both sites |
| | Weser estuary | 2.5–2.8 | |
| Se | Elbe estuary | 2.2–3.0 | Varying concentrations at both sites, no significant trend |
| | Weser estuary | 1.8–3.2 | |

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# 14 ISSUES REGARDING MARINE MAMMALS

## 14.1 Research Programme on Cause-Effect Relationships between Contaminants and Population-Level Effects in Seals

*Request*

There is no specific request; the development of a research programme on cause-effect relationships between contaminants and population-level effects in seals has been initiated given the current need to understand the effects of environmental contaminants on the health status of marine mammal populations.

*Source of the information presented*

The 1998 and 1999 reports of the Working Group on Marine Mammal Habitats (WGMMHA) and ACME deliberations.

*Status/background information*

The ACME noted that in 1998 WGMMHA reviewed available information on population-level effects of contaminants in marine mammals. WGMMHA concluded that chemical contaminants, particularly chlorobiphenyls, are likely to affect the health of certain marine mammal populations. However, the extent of this effect is unclear, despite some experiments linking contaminants to sub-cellular, cellular or systemic level effects. Suppression of fecundity rates and population growth have been reported for marine mammal populations resident in contaminated areas, but there is no well-defined cause-effect relationship linking specific contaminants to population-level effects. Therefore, in 1998 WGMMHA recommended the development of a research programme to explore these questions; this was endorsed by ICES.

At its meeting in 1999, WGMMHA prepared a project proposal that could be submitted for international funding. The objective of the project is to develop and expand biomarkers indicative of physiological responses of animals to contaminant exposure in controlled experiments. These developments will be applied to wild populations which occur over a gradient of pollution, and where good information is available on life history, population dynamics, and ecology.

The proposal considers a wide range of toxic compounds, but priority is given to 'classical' organochlorines (e.g., PCBs, DDTs, chlordanes) and brominated flame retardants. Research on biotransformation enzymes and other indicators of response to exposure, and on effects on the immune system, reproduction, and early development are identified as important fields of research. The field programme focuses on populations of harbour seals in the Wadden Sea, the Skagerrak, Moray Firth, and central Norway, with possible expansion to include grey seals in the Baltic Sea, Liverpool Bay (UK), central Norway, and Breidafjordur in Iceland, and ringed seals in the Baltic Sea and Svalbard area. A similar research effort on contaminants in cetaceans is under development by the Scientific Committee of the International Whaling Commission (IWC). Collaboration between the two research efforts would be highly desirable.

The ACME endorsed the above-mentioned proposal for a research programme on cause-effect relationships between contaminants and population-level effects in seals. The ACME welcomes the research plans by the IWC Scientific Committee and encourages collaboration between the research efforts planned by ICES and IWC.

## 14.2 Scientific Basis for Triennial Reviews of the Health Status of and Effects of Contaminants on Marine Mammals in the Baltic Sea

*Request*

Item 1 of the 1999 requests from the Helsinki Commission: to evaluate every third year the populations of seals and harbour porpoise in the Baltic Sea, including the size of the populations, distribution, migration, reproductive capacity, effects of contaminants and health status, and additional mortality owing to interactions with commercial fisheries (by-catch, intentional killing).

The next triennial review is scheduled for the year 2000.

*Source of the information presented*

The 1999 report of the Working Group on Marine Mammal Habitats (WGMMHA) and ACME deliberations.

*Status/background information*

New information on the development of the populations of the Baltic ringed seal *(Phoca hispida)* and grey seal *(Halichoerus grypus)* during the past century was presented to WGMMHA. Retrospective population simulations were based on uniquely detailed hunting records from the Nordic countries and published data on demographic parameters. Annual variations in the composition of catches by sex and age were taken into account. Grey seals decreased from 88 000–100 000 in the beginning of the century to approximately 4000 in the late 1970s. The Baltic ringed seal population decreased during the same period from 190 000–220 000 to approximately 5000. In the mid-1960s the remaining populations were afflicted by sterility, probably caused by organochlorines, which inhibited natural growth for at least 25 years. Thus, the decrease in seal numbers was a consequence of excessive hunting, but the low numbers at present are due to lowered fertility rates after 1965.

Concentrations of organochlorines in extractable fat from Baltic seals increased ten-fold during the latter half of the 1960s, after which time a significant increase was detected in the prevalence of pathological changes of reproductive organs, skeleton, and intestines. Data on the present growth rate of the ringed seal population (5 %) and pregnancy rates were used to model the flux in reproductive capacity in the past. It was found that, irrespective of which age structure was used in the model, the growth rate was negative in the period 1970–1984. Including detailed information on hunting mortality showed that the ringed seal population in the Bothnian Bay decreased from about 14 000 to less than 4000 during the period 1960 to 1986. Only very recently was a positive trend possible. Significant correlation was found between the reported reproductive capacity of ringed seals and the concentrations of PCBs in extractable fat from seals. The flux in reproductive capacity was also significantly correlated to changes in the concentrations of PCBs in the major prey item of ringed seals, the Baltic herring (*Clupea harengus*).

*Recommendations*

ICES encourages all Member Countries bordering the Baltic Sea to make available to ICES the relevant information on by-catches, population surveys, and health status of marine mammals in the Baltic prior to the review in the year 2000.

# 15 ISSUES RELATED TO MARICULTURE

## 15.1 Fishery and Mariculture Genetics Research

*Request*

There is no specific request; this is part of ongoing ICES work on genetic issues in fisheries and mariculture.

*Source of the information presented*

The 1998 and 1999 reports of the Working Group on the Application of Genetics in Fisheries and Mariculture (WGAGFM), papers presented at the Theme Session on the Use of Genetics in Aquaculture at the 1998 ICES Annual Science Conference, and ACME deliberations.

*Status/background information*

**Genetically Modified Organisms**

The only ICES protocol dealing with genetics is the ICES Code of Practice on the Introductions and Transfers of Marine Organisms, which is applicable to genetically modified organisms (GMOs). GMOs have been defined as organisms in which the genetic material has been modified by human technologies. The ICES recommended procedure for the consideration of the release of GMOs includes that the transgenic organisms must be reproductively sterile in order to minimize impacts on the genetic structure of natural populations.

The ACME took note of progress in transgenic studies on aquatic organisms, as summarized in Table 15.1.1. As progress in the development of transgenic organisms has been made, ICES is presently particularly concerned by the creation of transgenic salmon with growth hormone. To be certain of the sterility of the organisms, triploidy is generally not safe enough as animals can produce a small quantity of gametes. Strains of 100 % sterile fish could be obtained only by producing transgenic fish with gene inhibitions of gonadotropin-releasing hormone (GnRN) blocking the hypothalamic-pituitary-gonad axis of sexual maturation. Before that sterility can be clearly ascertained, the salmon must remain in enclosed land-based facilities in quarantine, avoiding sperm and ovule escape. Triploids can also be obtained by crossing diploids with tetraploids. In that case, the triploidy rate is 100 % with a guarantee of sterility. This technology is currently in practice for oysters, *Crassostrea gigas*. But guarantees must be obtained to keep the tetraploids in a quarantine system with assurance of gamete retention. If tetraploids are not under control, the risk of sterilization of the whole population (wild and cultivated) exists and has been estimated for some oyster populations.

**Fisheries Applications**

For fisheries purposes, molecular markers are commonly used. Microsatellites should be more fully exploited as markers for the study and management of fisheries populations. The new DNA technology permits an improved description of population structuring in species largely inaccessible using conventional protein markers, either due to limited polymorphism, or high sensitivity of allozymes to low levels of gene flow. Elucidation of the oceanographic, spatial, and temporal determinants of small-scale structuring would greatly facilitate our understanding of the variability in larval survival and recruitment dynamics, including the critical relationship between larval and adult populations. Studies on the extent to which juveniles, identified individually with microsatellites, become associated with returning adults to spawning grounds, for example, may provide significant data on the maintenance of seasonal spawning groups of fish, and the origin and persistence of meta-population structuring.

Attention should be given to methods for determining population structure and estimating gene flow, including examination of effective population size, determining the population of origin of a catch, and determining relatedness in open populations. It is urgent to involve both geneticists and modellers to test methods for these determinations.

**Marine Protected Areas (MPAs)**

Marine Protected Areas (MPAs) are being created for various conservation purposes. Now, measures are being proposed to create Marine Protected Areas to preserve marine genetic resources and diversity. The size and location of the MPA will vary according to many factors. The population size, the sex ratio, and the reproductive efforts are parameters which will define the characteristics of the MPA to avoid inbreeding.

The genetic resource management plans for MPAs should consider both the short-term preservation of endangered species as well as longer-term goals for the maintenance of genetic diversity within species in order to allow for adaptation to environmental change and for continued speciation that are needed to maintain evolutionary flexibility for the future.

Genetic risk assessment by fishery scientists and managers should be incorporated into MPA decision-making processes.

**Table 15.1.1.** Summary of transgenic studies on aquatic organisms from 1985–1997.

| Species | | Method | Promoter/Gene | Reference |
|---|---|---|---|---|
| Rainbow trout | *Oncorhynchus mykiss/* | Mi | mMT/rGH | MacLean and Talwar, 1984 |
| | *Salmo gairdneri* | Mi | SV40/hGH | Chourrout *et al.*, 1986 |
| | | Mi | mMT/rGH | MacLean *et al.*, 1987a |
| | | Mi | mMT/hGH | Guyomard *et al.*, 1989 |
| | | Mi | mMT/rGH | Penman *et al.*, 1988 |
| | | Mi | mMT/rGH | Penman *et al.*, 1990 |
| | | Mi | mMT/rGH | Penman *et al.*, 1991 |
| | | Mi | RSV/rtGH | Inoue *et al.*, 1993 |
| | | Mi | opAFP/csGH | Devlin *et al.*, 1995 |
| Cutthroat trout | *Oncorhynchus clarki* | Mi | opAFP/csGH | Devlin *et al.*, 1995 |
| Atlantic salmon | *Salmo salar* | Mi | wfAFP | Fletcher *et al.*, 1988 |
| | | Mi | mMT/CAT | McEvoy *et al.*, 1988 |
| | | Mi | mMT/hGH | Rokkones *et al.*, 1988 |
| | | Mi | wfAFP | Hew *et al.*, 1992 |
| | | Mi | opAFP/csGH | Du *et al.*, 1992 |
| Chinook salmon | *Oncorhynchus tshawytscha* | Mi | opAFP/csGH | Devlin *et al.*, 1995 |
| | | Sp | RSV/Gal | Sin *et al.*, 1993 |
| Coho salmon | *Oncorhynchus kisutch* | Mi | opAFP/csGH | Devlin *et al.*, 1995 |
| Tilapia | *Oreochromis niloticus* | Mi | mMT/hGH | Brem *et al.*, 1988 |
| | | Mi | mMT/rGH-CAT | Rahman and MacLean, 1992 |
| | | Mi | RSV/bGH | Phillips *et al.*, 1992 |
| | | Mi | mMT/rGH | Rahman and MacLean, 1992 |
| | | Mi | caBA/Gal | Alam *et al.*, 1996 |
| | | Mi | CMV/tiGH | Martinez *et al.*, 1996 |
| | | Mi | CMV/tiGH | Estrada *et al.*, 1996 |
| Medaka | *Oryzias latipes* | Mi | cCR | Ozato *et al.*, 1986 |
| | | Mi | RSV/CAT | Chong and Vielkind, 1989 |
| | | Mi | fLuc/luc | Tamiya *et al.*, 1990 |
| | | El | mMT/rGH | Inoue *et al.*, 1990 |
| | | Mi | mMT,vTK,rCCK,cBA/hGH | Lu *et al.*, 1992 |
| | | Mi | SV40,RSV,chMT,dhsp70/luc | Sato *et al.*, 1992 |
| | | Mi | rtMT/CAT | Kinoshita *et al.*, 1994 |
| | | Mi | medaka-actin/Gal | Takagi *et al.*, 1994 |
| | | Mi | RSV/Gal, CMV/Gal | Tsai *et al.*, 1995b |
| | | Mi | rtMT/CAT | Kinoshita *et al.*, 1996 |
| Goldfish | *Cassarius auratus* | Mi | mMT/hGH | Zhu *et al.*, 1985 |
| | | Mi | mMT/hGH | MacLean *et al.*, 1987b |
| | | Sp | RSV/Neo | Yoon *et al.*, 1990 |
| | | Mi | opAFP | Wang *et al.*, 1995 |
| Channel catfish | *Ictalurus punctatus* | Mi | mMT/rGH, RSV/rtGH, | |
| | | | RSV/csGH, RSV/rtV | Hayat *et al.*, 1991 |
| | | Mi | RSV/rtGH, RSV/csGH | Dunham *et al.*, 1992 |
| | | Mi, El | RSV/rtGH | Powers *et al.*, 1992 |
| Loach | *Misgurnus anguillicaudatus* | Mi | mMT/hGH | Zhu *et al.*, 1986 |
| (Oriental weatherfish) | | Sp | opAFP/csGH | Tsai *et al.*, 1995a |
| Common carp | *Cyprinus carpio* | Mi | RSV/rtGH | Chen *et al.*, 1989 |
| | | Mi | RSV/rtGH | Zhang *et al.*, 1990 |
| | | Mi | mMT/rGH, RSV/rtGH | |
| | | | RSV/csGH, RSV/rtV | Hayat *et al.*, 1991 |
| | | Mi, El | RSV/rtGH | Powers *et al.*, 1992 |
| | | Mi | RSV/rtGH | Chen *et al.*, 1993 |
| | | Mi | caBA/csGH | Moav *et al.*, 1995 |
| Northern pike | *Esox lucius* | Mi | RSV/bGH, RSV/rtV | Gross *et al.*, 1992 |
| Pacific oyster | *Crassostrea gigas* | Bo | dhsp70/luc, CMV/luc | Cadoret *et al.*, 1997 |
| Dwarf surfclam | *Mulinia lateralis* | El | PPRV(Gal) | Chen *et al.*, 1996 |
| | | El | PPRV(Gal) | Kan *et al.*, 1996 |
| Abalone | *Haliotis rufescens* | El | d-actin/Gal | Powers *et al.*, 1995 |
| | | El | aBA/luc,Gal | Powers *et al.*, 1996 |
| | | Sp | opAFP/CAT | Tsai *et al.*, 1997 |
| Artemia | *Artemia franciscana* | Bo | dhsp70/luc | Gendreau *et al.*, 1995 |

Explanatory Notes for Table 15.1.1.

Studies published prior to 1992 were compiled from Chen and Powers (1990), Brem (1993), and Beaumont (1994). Studies published from 1992 to 1997 were compiled from ASFA (Aquatic Sciences and Fisheries Abstracts), Biological Abstracts, and MedLine databases.

**Methods**: Mi = microinjection; El = electroporation; Sp = sperm vector; Bo = particle bombardment.

**Promoters**: MT = metallothionein promoter: mMT = mouse MT, chMT = chinese hamster MT, rtMT = rainbow trout MT; BA = beta-actin: cBA = chicken BA, caBA = carp BA, aBA = abalone BA; SV40 = SV40 promoter; opAFP = ocean pout antifreeze protein promoter; wfAFP = winter flounder antifreeze promoter and protein; RSV = Rous sarcoma virus long terminal repeat promoter; CMV = promoter of the immediate early gene of the human cytomegalovirus; cCR = chicken crystallin promoter and gene; fLuc = firefly luciferase; vTK = viral thymidine kinase; rCCK = rat cholecystokinin; dhsp70 = drosophila heat-shock protein 70; d-actin = drosphila actin promoter; PPRV(Gal) = pantropic pseudotyped retroviral vector containing Gal.

**Genes**: GH = growth hormone: rGH = rat GH, hGH = human GH, bGH = bovine GH, rtGH = rainbow trout GH, csGH = chinook salmon GH, coGH = coho salmon GH, tiGH = tilapia GH; CAT = bacterial chloramphenicol acetyltransferase; opAFP = ocean pout antifreeze protein; wfAFP = winter flounder antifreeze protein; rtV = rainbow trout vitellogenin; Neo = neomycin resistance; Gal = beta galactosidase; luc = firefly luciferase.

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## 15.2 Environmental Interactions of Mariculture

*Request*

There is no specific request; this is part of the continuing ICES work on environmental issues relating to mariculture.

*Source of the information presented*

The 1999 report of the Working Group on Environmental Interactions of Mariculture (WGEIM) and ACME deliberations.

*Status/background information*

The ACME took note of information from WGEIM, as summarized below.

**Sea lice control and the use of chemicals**

Sea lice control has remained a general problem in commercial salmon cultivation. Additionally, declines in wild salmonid stocks noted in Ireland, Scotland, Norway, and Canada have been linked by some investigators with sea lice populations at salmon farms. Ireland has instigated a compulsory sea lice monitoring programme at fish farms and has installed counters on many rivers. Lice control was largely through the use of chemicals including cypermethrin and ivermectin. Norway is undertaking additional studies of basic louse biology, and there are proposals to exclude fish farms from some fjords, close rivers to fishing, and ban fish farming in some regions (e.g., eastern Finmark). There is increasing pressure against the use of medicines as feed additives to prevent the possible transfer of medicine residues to wild fish eaten by consumers. There is particular pressure on diflubenzuron, which has been reported to degrade to a carcinogen (*p*-chloro-aniline). There is pressure in Scotland for the introduction of an effective and environmentally acceptable approach to sea lice control.

The tendency towards increased pressure against the use of chemicals (including therapeutants) at fish farms was noted. In Norway, the use of in-feed treatments for disease or parasite infection is likely to become more difficult to sustain because of the possible transfer of such chemicals to wild fish feeding on excess pellets and faeces. Reports of antibiotics found in wild fish near Norwegian fish farms for about ten years have been very influential.

In Scotland, a new field programme of investigations at salmon farms (Post-Authorisation Monitoring Programme (PAMP)) into the possible community-level effects of sea lice treatment chemicals has been initiated by a consortium of funding agencies. The funding agencies include governmental environment and fisheries departments, conservation organizations, the Scottish Environmental Protection Agency, the Scottish Salmon Growers Association, and individual salmon farming companies, with support from the relevant pharmaceutical companies.

Sea lice treatment chemicals are categorized as medicines under UK (and EU) legislation. Before commercial preparations for the control of sea lice in mariculture can be placed on the market, it is necessary for the manufacturer to obtain a Marketing Authorisation (MA) under UK and EU legislation.

Applications for MAs are assessed under the UK Medicines Act which requires the licensing authority (Secretaries of State for Agriculture, acting on the advice of the independent Veterinary Products Committee (VPC), supported by the Veterinary Medicines Directorate (VMD) of the Ministry of Agriculture, Fisheries and Food (MAFF)) to take into account factors relating to:

a) the pharmaceutical quality of the product;

b) the efficacy of the product;

c) the safety of the product to consumers, operators, and the environment.

Safety to the environment is assessed from data generated in a tiered series of tests ranging from simple determinations of physico-chemical properties to multispecies mesocosm studies, and modelling of the behaviour and fate of the substance in the environment. In most cases, the data required by VPC/VMD stop short of experimental mesocosm studies, but include acute and chronic toxicity tests for single species, and observations of the behaviour and biological effects of the product in field trials.

This information, together with the establishment of a Maximum Residue Level by the European Medicines Agency, forms the core of the safety package in an application for an MA.

Concern has been expressed by conservation organizations, and others in the UK, that the medicines assessment procedure does not take into account possible long-term environmental effects, particularly at community and higher levels. There is very little, if any, scientific evidence to suggest that significant effects have arisen from the use of sea lice treatments (primarily dichlorvos and hydrogen peroxide) to date. However, studies at the community level are not easy to undertake, and only a few studies of community effects of sea lice treatments have been attempted. It was in recognition of this problem that the Post-Authorisation Monitoring Programme was initiated to study community-level effects of the use of sea lice treatments at salmon farms in Scotland.

The ACME considered that, while it is necessary to undertake appropriate monitoring activities at fish farms, the long-term solution to sea lice problems would lie in the area of control of the lice at levels which posed no direct or indirect threat to the environment. It was essential to continue basic biological and other research into sea lice to identify effective control strategies and techniques.

Some other cultivation systems have been shown, or were likely, to have the potential to reduce lice levels. Pump-ashore tank sites generally did not have lice problems. It was suggested that closed cage systems with pumped water intakes from depth might avoid exposing the fish to 'infective' life stages of lice. A few commercial-scale systems were in use in Norway and under trial in Sweden. The main purpose of the systems was to control the temperature of the water in the cages (usually to increase temperatures in winter). It was noted that such systems were normally rather expensive to install and operate, but that they might have application in particularly sensitive areas, and they could offer the potential for the confinement and treatment of effluents. Even with more traditional cage designs, there could be the possibility of retaining treatment chemicals in baths and using swim-over techniques to treat the cultured fish.

Wrasse were used at many fish farms in Norway, but were less common in Scotland. This difference was probably linked to the availability (and cost) of the wild-caught wrasse from limited stocks. It has been shown that wrasse are only effective on small fish, and that they are largely ineffective in the winter and they can carry salmonid pathogens.

**Non-native species introduced through aquaculture**

The ACME took note of several cases of accidental introductions of non-native species that have occurred in association with the movement of live organisms for mariculture purposes (see also Section 11.1, above).

It was reported that near the island of Sylt in the North Sea, Pacific oysters, *Crassostrea gigas*, have been grown on trestles for more than ten years, and this cultivation was reliant upon imported seed from certified hatcheries in Ireland. Seed was normally taken directly from the hatchery to Sylt. However, occasionally it has been necessary for the seed to be stored outside the hatchery for a few days before export. This led to some growth of fouling organisms on the seed and the importation to the Sylt area of the Wadden Sea of the seaweeds *Sargassum muticum* and *Ascophyllum nodosum* and the seasquirts *Aplidium nordmanni* (an ascidian) and *Styela clava* (which originated in the Eastern Pacific).

*Crassostrea gigas* now seems to be established (probably permanently) in the wild in the Wadden Sea. Spatfalls occurred in 1991 and 1994, and the population was estimated at 1 000 000 in summer 1995, of which 66 % survived the subsequent severe winter. The animals grew to 50–80 mm in two years, and the size structure of the population suggested the survival of some animals from the 1991 spatfall. The animals were found as an epibiont on densely packed mussel beds, from which the normal common macroalgae were absent. *Crassostrea* therefore does not occupy the niche of the regionally extinct flat oyster, *Ostrea edulis*, but constitutes an r-selected species invading a crowded community in an undisturbed habitat.

It was reported that more than 50 % of the macroalgal species in the Thau lagoon in France have been introduced, almost certainly in association with imports of oysters for mariculture purposes.

From Canada, the occurrence of juvenile Atlantic salmon in the Tsitika River, British Columbia was reported. The presence of two age classes of juvenile Atlantic salmon in a river which hosts no hatcheries containing that species suggested that Atlantic salmon had successfully spawned in the river in 1996 and 1997. There was some doubt as to the numbers of juveniles present, the numbers of other salmonids present, and the geographical distribution of the juveniles. The presence of these juveniles necessitated an evaluation of their ability to establish an endemic population and compete with local salmonid species. Concern was constrained by the fact that there had been numerous introductions of as many as 500 000 to one million eggs or fry to other river systems and no recorded returns from the smolts that went to sea. In addition, limited genetic evidence collected from this occurrence suggests that all the fish may have been the result of a single pair of fish mating in 1996 and another pair of fish in 1997. Further, even at the most extreme estimate of the abundance of Atlantic salmon fry, they were very rare when compared to local salmonid species.

The ACME noted the increasing recognition of the risk of inadvertent transfers of alien species when moving marine organisms from place to place. In some areas, aquaculture is dependent on the importation of juveniles or other life stages. The development of local broodstocks and sources of juveniles is clearly a method of reducing this risk. While some introductions may present hazards to the wider environment in the receiving area, other importations can present direct hazards to aquaculture. Pests and diseases carried in the transferred organisms are a clear example. In addition, the transfer of toxic algae, with imported animals or by other means such as ballast water, represents a significant risk factor. Norway has recently experienced a bloom of *Chatonella*, thought to have been imported in ballast water, while there have been occurrences of a previously unknown toxin in Ireland and a similar experience in the Thau lagoon in France.

**Other introduced pests**

It was reported that oyster producers in some parts of Atlantic France were experiencing considerable increases in the populations of *Crepidula fornicata* (slipper limpet). *Crepidula* was introduced to the area from North America about fifty years ago, but has recently increased in numbers such that they have now attained 30–40 % of the standing stock of oysters in some areas. The *Crepidula* was expanding even in areas where oyster stocks were too high, and were competing for the energy resources. Growers were trying to remove them by trawling, but this was costly, and there was no significant market for the *Crepidula*.

# 16 MARINE HABITAT CLASSIFICATION AND MAPPING

*Request*

There is no specific request; habitat classification is a new field within ICES.

*Source of the information presented*

Terms of reference for an OSPAR/ICES/EEA Workshop on Habitat Classification and Biogeographic Regions, a 1999 progress report from the Study Group on Marine Habitat Mapping (SGMHM), and ACME deliberations.

*Status/background information*

Marine habitat classification and habitat mapping are essential to obtain a good, basic understanding of the natural conditions present in the coastal zone, and to answer questions as to whether habitats are at risk of being damaged or lost as a result of human impacts on the marine environment. Against this background, ICES established a Study Group on Marine Habitat Mapping (SGMHM), under the Marine Habitat Committee, at the 1998 Annual Science Conference. The main tasks of SGMHM are to explore whether a habitat classification system can be developed for the ICES area. Habitat classification is a new field within ICES, but a great deal of activity in this field is already going on outside ICES. Therefore, it was felt necessary that wide participation of relevant experts would be ensured, e.g., through cooperation with the OSPAR Working Group on Impacts on the Marine Environment (IMPACT) or another international group.

In its work programme, SGMHM will place emphasis on the following activities:

a) considering whether the EUNIS (European Nature Information System) BioMar classification system currently under construction in the European Environment Agency (EEA) is adequate to the needs of ICES;

b) if it is useful, determining how it can be extended or tailored to the needs of ICES;

c) proposing how the contribution of ICES to developing a habitat classification system for the ICES area can be organized.

The conduct of a joint OSPAR/ICES/EEA Workshop on Habitat Classification and Biogeographic Regions, in Oban, Scotland from 6–10 September 1999, was noted. This Workshop aims at the development of a habitat classification (inventory) for the OSPAR area, and will shed light on the question of whether the BioMar classification (developed for the inshore areas around Ireland and the UK) is useful to the needs of ICES for application on a broader geographical basis.

Some conclusions of this workshop were later noted by the ACME, at its September 1999 meeting. It was noted that the EEA EUNIS classification was discussed with regard to its applicability to the OSPAR and ICES areas of responsibility. This applicability was discussed in relation to three types of habitats: rocky, sediment, and deep-sea habitats.

In terms of rocky habitats, it was determined that the current classification is acceptable to level 3, but that new types may be required for level 4, especially for deep rock. For sediment habitats, there had been considerable discussion of the descriptors 'infralittoral' and 'circalittoral'. It was considered too difficult to define these categories clearly, so no distinction will be made. Revised definitions were proposed for rock and sediment in the classification. Regarding deep-sea habitats, it was concluded that the current classification was poorly developed for such types of habitat. A more detailed and restructured classification was developed, including preparation of some habitat descriptions.

The ACME agreed that, as it is recognized that wider expert participation from outside ICES is needed to link effectively to initiatives already started in the field of habitat classification and mapping, active support should be given to this joint initiative.

## 17 ASSESSMENT TOOLS

### 17.1 Environmental Indicators

*Request*

There is no specific request; this is part of new work to consider the development of scientifically sound ways of characterizing the situation or condition in the marine environment that can be understood by policymakers and managers.

*Source of the information presented*

The 1999 report of the Working Group on Environmental Assessment and Monitoring Strategies (WGEAMS), a discussion paper entitled 'Development of marine environmental quality indicators', and ACME deliberations.

*Status/background information*

The ACME is aware that there is a need to establish consistent environmental response criteria or indicators that are easily understood and accepted by policymakers and managers. Presently, a variety of indicators appear in environmental reports of national and international bodies, which limits comparisons among areas and over time.

WGEAMS considered this issue on the basis of a discussion paper entitled 'Development of marine environmental quality indicators', that was prepared by E. Andrulewicz (Poland). This discussion paper proposes the development of an environmental quality 'toolbox' of methods for scientists to provide advice to decision makers for the protection and sustainable use of the marine environment. The toolbox proposed contains the following three components:

1) a *marine classification system* that identifies the quality and status of the environment;
2) the classification system is based on *indicators* that measure the environmental status and trends;
3) a set of *reference values* (background, present, target) that can be used to assess environmental degradation and trends.

This discussion paper presents a list of nine principal problems caused by human activities that affect the marine environment and proposes a set of indicators for each of these problems:

1) eutrophication;
2) chemical contamination (persistent harmful substances);
3) petroleum hydrocarbon contamination;
4) artificial radionuclides;
5) exploitation of fish;
6) exploitation of mineral resources;
7) sanitary state (bacteriological contamination);
8) coastal degradation;
9) threats to marine biodiversity.

The set of indicators proposed is based on the 'core indicators' from the OECD (1993) report and others proposed by the author as a starting point for further development. The indicators are based on the pressure–state–response framework:

- pressure—general social and economic indicators of wealth, population, and other demographics, and specific indicators of natural resource use and consumption;
- state—measures of current environmental quality; and
- response—government policies and regulatory efforts, and societal responses, either collectively or individually.

Other efforts at indicator development include the following:

1) The European Environment Agency (EEA) has adopted a similar framework/causality chain: driving forces–pressure–state–impact–response (DPSIR). The European Topic Centre on Marine and Coastal Environment has developed a set of indicators based on the DPSIR pressure–state–response framework for the EEA (ETC, 1997) and a pilot study was conducted that focused on the following issues:
   - pollution (heavy metals);
   - eutrophication/saprobiation;
   - fishing;
   - loss and degradation of habitats;
   - groundwater extraction;
   - climate change.

   The pilot project developed a qualitatively scaled system based on actual data. Data for coastal areas of the Netherlands were used for most of the work. The lack of comparable national-level data from EU countries was an obvious shortcoming of the analysis, particularly in regard to time-series data. Nevertheless, the pilot project is an early attempt at an indicator system simulation.

2) The European Environmental Pressure Indices Project has developed sets of indicators for ten policy fields, one of which is Marine Environment and Coastal Zones. The six indicators for this policy field are: eutrophication, overfishing, development along shore, discharges of heavy metals, oil pollution at coast and at sea, and discharges of halogenated organic compounds. The indicator sets are being aggregated into pressure indices, one for each policy field (EUROSTAT, 1998).

3) The Nordic Council of Ministers has released the document 'Indicators of the Environment in the Nordic Countries'.

4) The UNEP Global Environmental Outlook project is also using the pressure–state–response framework.

5) The Food and Agriculture Organisation (FAO) of the United Nations has developed socio-economic indicators for agriculture, including general economic categories and agriculture-specific practices (Borysiewicz, 1997) and fisheries indicators.

6) OSPAR and ICES co-sponsored a Workshop on Background Concentrations (OSPAR, 1997). ICES accepted the definitions of background concentrations and reference values.

7) The World Resources Institute is in the process of developing indicators for sustainable development (Project SCOPE) for the UN Commission on Sustainable Development.

8) The Workshop on Eco-efficiency, Resource Productivity and Innovation, organized by the European Environment Agency in October 1998 (EEA, 1998), introduced the concepts of 'eco-efficiency', 'eco-intensity', and 'resource productivity' and presented indicators for these concepts.

9) The Workshop on Ecological Quality Objectives for the North Sea, held in Scheveningen, the Netherlands in September 1999, was arranged by Norway and the Netherlands in cooperation with the North Sea Conference Secretariat and co-sponsored by ICES (see Section 17.2, below).

*Need for further research or additional data*

The ACME noted that the strengths and limitations of the individual indicators are directly related to the knowledge of the systems and processes in which they were developed and to which they are applied. The more that is known about a system or process, the easier it is to choose appropriate indicators and apply them correctly. Errors in choosing indicators and uncertainties in applying them seriously affect assessment results and interpretation. The ACME also noted that the understanding of ecosystem processes is not consistent among systems so the selection of indicators, and the confidence in their ability to measure meaningful change, will not be consistent. Because a complete understanding is lacking, indicators may perform differently between systems and between components of a system.

The ACME noted that there is an urgent need to establish criteria or indicators for environmental responses that are easily understood and accepted by policymakers and managers. The ACME recognized the difficulties in translating scientific advice into common language, and especially the difficulties in understanding indicators. The ACME considered that scientific indicators could be translated into metrics that are more easily grasped by the public. While the indicators may lose some scientific exactness in this translation, the loss would be more than offset by the gain in public understanding and acceptance. Because policymakers provide funding for research, the ACME felt it important that scientific advice be formulated in an understandable way.

The problems of establishing simple, acceptable, scientifically-based indicators for describing positive and negative environmental changes and the requirements for communicating scientific advice to policymakers and managers need to be addressed by relevant ICES Working Groups, and reconsidered by ACME when progress has been made.

**References**

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OSPAR. 1997. OSPAR/ICES Workshop on the Overall Evaluation and Update of Background/Reference Concentrations for Nutrients and for Contaminants in Sea Water, Biota and Sediments. OSPAR Commission, London.

UNCED. 1987. World Commission on Environment and Development (Brundtland Commission). Our Common Future. Oxford University Press. 400 pp.

## 17.2 Ecological Quality Objectives for the North Sea

The ACME was informed that a Workshop on Ecological Quality Objectives for the North Sea would be held in Scheveningen, the Netherlands, from 1–3 September 1999. The Workshop is arranged by Norway and the Netherlands in cooperation with the North Sea Conference Secretariat, and is co-sponsored by ICES. It is a continuation of the work initiated by the Third North Sea Conference in 1990, the North Sea Task Force, and subsequently the OSPAR Commission, on the development of ecological objectives for the North Sea and its coastal waters. The Workshop aims at identifying parameters for ecological quality objectives (EcoQOs) based on the framework methodology already developed in OSPAR.

A working document was provided to ACME that was intended for distribution as a background document for the Workshop. The working document was in draft form, and comments and guidance were invited for further development of the document in advance of the Workshop. It contained four main sections: (i) an introductory discussion of the framework for the development of an ecosystem approach and Ecological Quality Objectives, (ii) an overview of general objectives and principles of relevant international agreements, (iii) a description of principles for EcoQO development, including aspects on fisheries, eutrophication, hazardous substances, and sand and gravel extraction, as well as examples mainly taken from recent developments in the Netherlands, and (iv) a list of proposals for new EcoQOs for the North Sea.

The ACME welcomed the initiative on the development of a basis for the management of marine areas based on an ecosystem approach. It was noted that this was very timely in relation to the strategic planning process currently under way in ICES, where one of the ICES goals will be the compilation of a set of indices (the development of a 'toolbox') to be used for assessing the state of an ecosystem as a basis for ecosystem management.

It was also noted that different EcoQOs may be chosen for different ecosystems, because our understanding of relevant processes is not comparable between systems (i.e., some systems are better understood than others), and this would be reflected in confidence in the ability of indicators to meaningfully measure change. Indicators may perform differently between systems and/or between components of a system. It would, however, not be advisable to expect to attain complete understanding of the natural variability in a system before choosing such indicators.

The ACME noted that the stated objective of the Workshop is to identify parameters for EcoQOS, but not necessarily limits for these parameters. Limits are discussed in the context of fisheries, but the report could contain a general discussion of limits and provide examples in additional areas (e.g., hazardous substances and reproductive success, nutrients and eutrophication). It could be advantageous to include a discussion of different reference values, for example, (i) background values as indicators in the absence of anthropogenic effects, (ii) recent values, measured under the current environmental situation, and (iii) target/limit values as acceptable environmental targets.

In this document, effects were classified into direct, indirect and knock-on. Direct effects were characterized by a functional relationship between dose and immediate effect, indirect effects are delayed responses in the system that can still be correlated with the primary effects, and knock-on effects are characterized by a longer chain of processes and greater delays, thus allowing for more complex relationships than indirect effects. The discussion of this classification was considered to be very useful to managers and policymakers. While presented in a fisheries context, it could be useful to provide a similar classification for other issues. The classification scheme could give managers and policymakers a more realistic expectation in relation to EcoQO indicators.

## 18 GLOBAL PROGRAMMES

### 18.1 Global Ocean Ecosystem Dynamics (GLOBEC) Programme

*Request*

There is no specific request; this is part of continuing ICES work on marine issues.

*Source of the information presented*

The reports of seven ICES/GLOBEC Workshops and meetings, as listed in the reference list, and ACME deliberations.

*Status/background information*

The Global Ocean Ecosystem Dynamics (GLOBEC) programme has a number of clearly identifiable elements and products and also many related activities (such as Theme Sessions at ICES Annual Science Conferences, the Working Group on Zooplankton Ecology (WGZE), the Trans-Atlantic Study of *Calanus finmarchichus* (TASC) programme and Symposium), which may or may not be included, depending on the context.

Six ICES/GLOBEC reports were presented at the 1998 Annual Science Conference, including reports of workshops on predictability of ocean climate, the application of environmental data in stock assessments, and the consequence of ocean climate variability on Northwest Atlantic gadoid populations during the 1960s and 1970s. A workshop in March 1999 explored the consequence of ocean climate on North Sea gadoid populations during the 1960s and 1970s (ICES, 1999). The workshops provided an excellent compilation and analysis of the physical variables affecting fish as well as of the temporal and geographical patterns of change in ocean climate. The workshops considered many processes by which ocean climate variability may affect gadoid stocks; some could be ruled out but others, which link physical variables to changes in growth and recruitment, were supported. The next step will be to explore particular processes in more detail and to use improved data sets in order to test the validity of hypothesized relationships. Some of these may then be developed as prognostic tools for improved short-term assessments and to estimate the environmental component of long-term fish stock fluctuations. The latter has considerable consequences when evaluating long-term sustainability, anthropogenic effects, and the application of the precautionary approach against a background of variability in ecosystem and fisheries productivity.

The ICES/GLOBEC Newsletters provide further information about progress at regional and national levels and are intended to help with coordination and planning. They are regularly circulated to about 250 scientists and can all be viewed on the web pages http://www.ices.dk/globec/globec.htm. The International GLOBEC Newsletter also carries much information about the development of the programme and includes ICES events, such as Theme Sessions and Symposia; the address is http://www1.npm.ac.uk/globec/index.htm.

GLOBEC is the only current international global change programme in which ICES plays a major part, as a regional partner. The plan for future work by the ICES Working Group on Cod and Climate Change (WGCCC) is included in the International GLOBEC Implementation Plan.

**References**


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ICES. 1998. Report of the ICES/GLOBEC Workshop on Prediction and Decadal-Scale Ocean Climate Fluctuations of the North Atlantic. ICES CM 1998/C:14.

ICES. 1999. Report of the Workshop on Gadoid Stocks in the North Sea during the 1960s and 1970s, the Fourth ICES/GLOBEC Backward-Facing Workshop. ICES CM 1999/C:15.


### 18.2 Global Ocean Observing System (GOOS)

*Request*

There is no specific request; this is part of continuing ICES work on marine monitoring and assessment.

*Source of the information presented*

The report of the 1999 Workshop on GOOS and ACME deliberations.

*Status/background information*

The ICES Workshop on GOOS made substantial progress in developing a framework for ICES involvement in the Global Ocean Observing System (GOOS). The Workshop benefited greatly from the participation of representatives from GOOS and EuroGOOS. Representatives attended from Canada, Germany, Norway, Sweden, UK, and USA, and written reports were submitted from Iceland and Spain. Clearly there is a broad base of interest among ICES Member Countries regarding an ICES involvement in GOOS. In addition, the North Sea Conference Secretariat participated in the Workshop in order to provide a background for defining future North Sea monitoring requirements.

The Workshop noted that 'operational oceanography' is central to GOOS, but this term is subject to various interpretations. The Workshop adopted the EuroGOOS definition for 'operational oceanography', namely,

'Operational oceanography is the activity of routinely making, disseminating, and interpreting measurements of the seas and oceans and atmosphere so as to:

- provide continuous forecasts of the future condition of the sea for as far ahead as possible (*Forecast*);
- provide the most usefully accurate description of the present state of the sea including living resources (*Nowcast*);
- assemble climatic long-term data sets which will provide data for description of past states, and time series showing trends and changes (*Hindcast*).'

ICES initiatives of potential relevance to GOOS were reviewed. It was concluded that the following should play a central role in an ICES-GOOS:

- the Annual Ocean Climate Status Summary (product of the Working Group on Oceanic Hydrography (WGOH) and based on a series of fifty standard hydrographic sections and stations from the entire North Atlantic which are maintained by ICES Member Countries) (see Section 8.4.1, above);
- phycotoxin decadal maps (product of the ICES/IOC Working Group on Harmful Algal Bloom Dynamics (WGHABD), see Section 8.4.2, above, and Annex 2);
- zooplankton monitoring (product of the Working Group on Zooplankton Ecology (WGZE)).

The Workshop was also informed about the ICES International Bottom Trawl Surveys (IBTS), which were immediately recognized as a potential candidate for the Initial Observing System (IOS) of GOOS, as the survey appeared to conform with 'GOOS Principles'. The GOOS Living Marine Resource (LMR) Panel was asked to confirm the Workshop's view. The Panel later confirmed this, stating that the operational nature of these surveys, the element of regional cooperation, and the survey's collection of oceanographic monitoring data along with fisheries data, were consistent with the observing programme envisioned for LMR GOOS. The Panel invited ICES endorsement of this proposal.

Having reviewed the above-mentioned existing activities, the Workshop then identified an ICES-GOOS as initially comprising two main elements:

1) an Atlantic component focusing on ocean climate, consisting of an enhanced ICES standard sections and stations, climate databases, and climate summary publications such as the Ocean Climate Status Report, as a cooperative effort among ICES Member Countries, including Canada and the USA, and EuroGOOS;
2) an ICES regional GOOS system on an appropriate time scale for the North Sea, focusing on ecosystem dynamics with special emphasis on the needs for improving the management of fish stocks.

For the future, the Workshop saw an important role for ICES in helping to incorporate the scientific management of fisheries into the framework of GOOS. In the ICES area, the most important data originators are still predominantly fisheries research institutes. Additionally, fisheries management probably represents one of the most important customers for GOOS. Consequently, active ICES participation in GOOS may result in putting more emphasis on fisheries and fisheries management into the GOOS concept. To date, GOOS activities appear to be focused on a free-market approach to ocean observing systems rather than management of a common resource in a sustainable way.

Based on the above, the Workshop identified three main elements of an implementation plan for ICES-GOOS which should become the responsibility of a reconstituted Steering Group on GOOS (SGGOOS). These are:

1) Global and regional linkage (e.g., ICES working closely with IOC (ICES/IOC SGGOOS, including members who are national GOOS representatives, regional GOOS representatives, etc.));
2) ICES Ocean Observing System (I-OOS) (e.g., propose relevant ICES monitoring activities, ICES Ocean Observing System, coordination of timely data management, ICES Climate Status Summary);
3) Regional ICES-GOOS component for the North Sea (in cooperation with EuroGOOS) to establish a coordinated and harmonized observation network, initially in the North Sea, in conformity with GOOS principles.

The Steering Group will address ways of promoting the role of ICES in GOOS, taking into account input from ICES Advisory and Science Committees.

# 19 DATA HANDLING

## 19.1 Activities of the ICES Environmental Data Centre

*Request*

Item 5 of the 1999 Work Programme from the OSPAR Commission: to carry out data handling activities relating to:

5.1 contaminant concentrations in biota and sediments;

5.2 measurements of biological effects.

Contract from the Arctic Monitoring and Assessment Programme (AMAP) to serve as Thematic Data Centre for the marine component during 1998 and 1999.

Contract from the Helsinki Commission (HELCOM) to serve as Thematic Data Centre for Baltic Monitoring Programme data for a three-year period beginning on 1 July 1998.

*Source of the information presented*

Progress report from the ICES Environmental Data Centre and ACME deliberations.

### 19.1.1 Databases on contaminants in marine media, biological effects of contaminants, and fish disease

The ACME took note of the information presented by the ICES Secretariat on the handling of data on contaminants in biota, sediments, and sea water, as well as data on fish disease. In this context, the ACME noted that, at present, the ICES Environmental Data Bank includes the following components:

1) contaminants in marine invertebrates, fish, birds, and mammals (approximately 330 000 records);
2) contaminants in sea water (approximately 280 000 records);
3) contaminants in marine sediments (approximately 80 000 records);
4) biological effects of contaminants (approximately 4000 records);
5) fish disease prevalence data (approximately 80 000 records).

The annual flow of data into the ICES Environmental Data Centre is approximately 25 000 records for each of the first two components and 5000 records for the third component. The flow of new data into the biological effects database is small. The data holdings according to year are illustrated in Figures 19.1.1 to 19.1.4. It should be noted that the small number of data records from the most recent years does not necessarily represent a downward trend in data submissions, but merely illustrates the natural time lag between the actual monitoring and the submission of data to the ICES Environmental Data Centre. No figure has been shown for the annual holdings of data on fish disease prevalence owing to the difficulties of extracting this information in a comparable way from this separate database.

Since mid-1998, ICES has served as the thematic data centre (TDC) for HELCOM under a three-year contract. ICES has been contracted to function as the TDC for the Baltic Monitoring Programme (oceanographic, biological and contaminants) data. This activity, as well as the intended inclusion of older data, is expected to increase the flow of data substantially.

### 19.1.2 Quality Assurance Database

The ICES Quality Assurance Database consists of the following components:

1) composition of reference materials;
2) written documentation from the laboratories on monitoring activities, analytical methods, etc;
3) data generated via intercomparison exercises.

The first component currently consists of descriptions of approximately 150 materials with their referenced and/or recommended composition. The second component consists of approximately 250 documents. Countries and/or laboratories reporting data to the ICES Environmental Data Centre are requested to supply additional written information about, e.g., sampling and analytical procedures, but major gaps have been identified in this compilation. The OSPAR procedure has recently been strengthened in this respect and this is expected to result in a more stable flow of information.

The component on intercomparison exercises is based on two major sources of information: the ICES intercomparison exercises and exercises carried out under QUASIMEME I and II. The development of the component based on the ICES intercomparison exercises is hampered by the fact that, in most cases, only paper versions of the reports of the exercises exist. These paper copies are being digitized at the speed that staff resources permit. At present, five of 49 exercises have been digitized.

**Figure 19.1.1.** Number of records of data on contaminants in biota in the ICES Environmental Data Centre according to year of sampling.

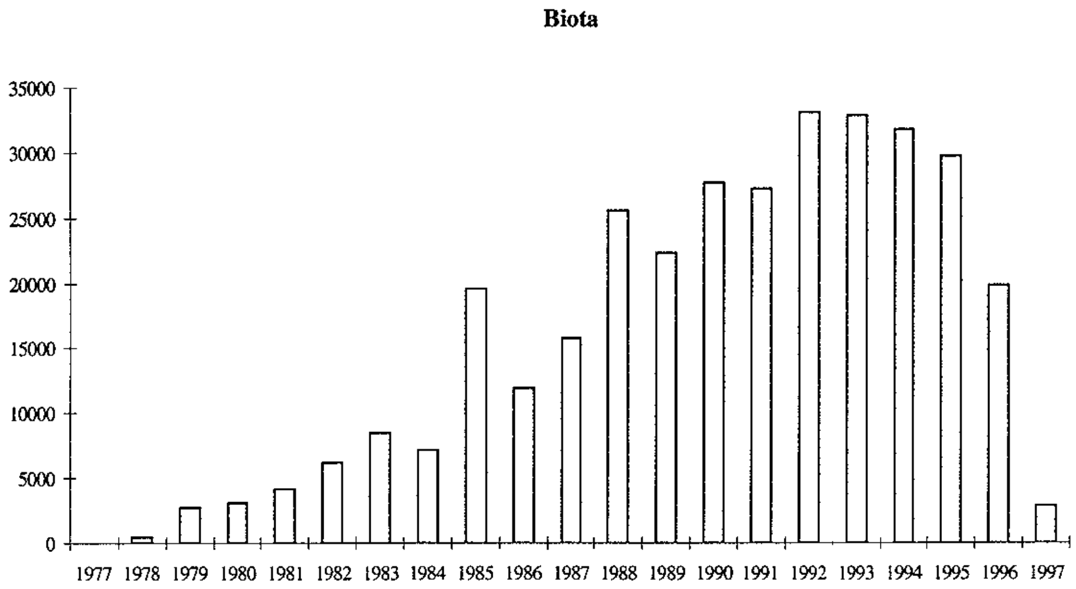



**Figure 19.1.2.** Number of records of data on contaminants in sea water in the ICES Environmental Data Centre according to year of sampling.

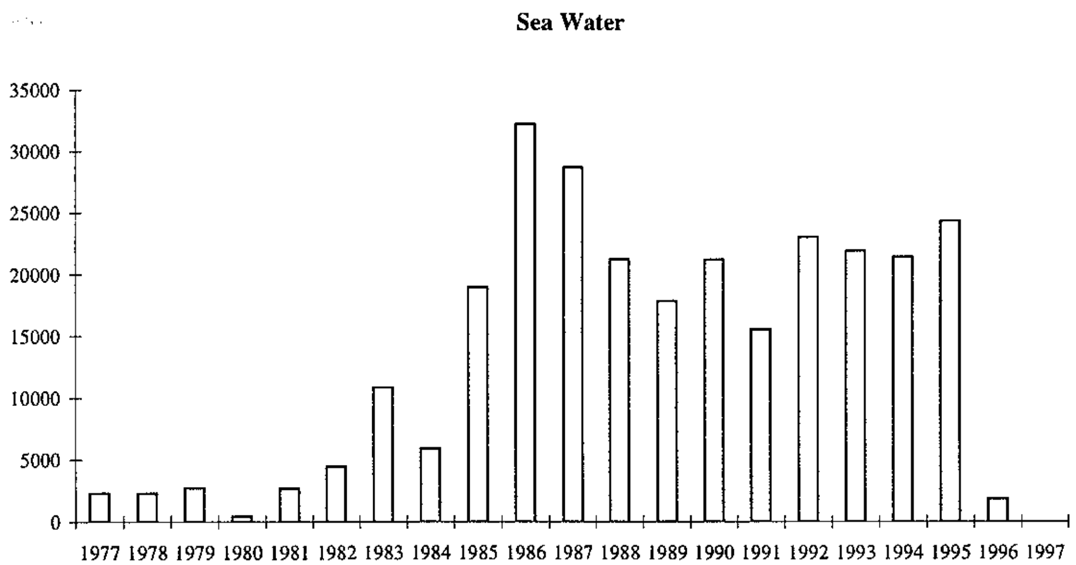


**Figure 19.1.3.** Number of records of data on contaminants in sediments in the ICES Environmental Data Centre according to year of sampling.

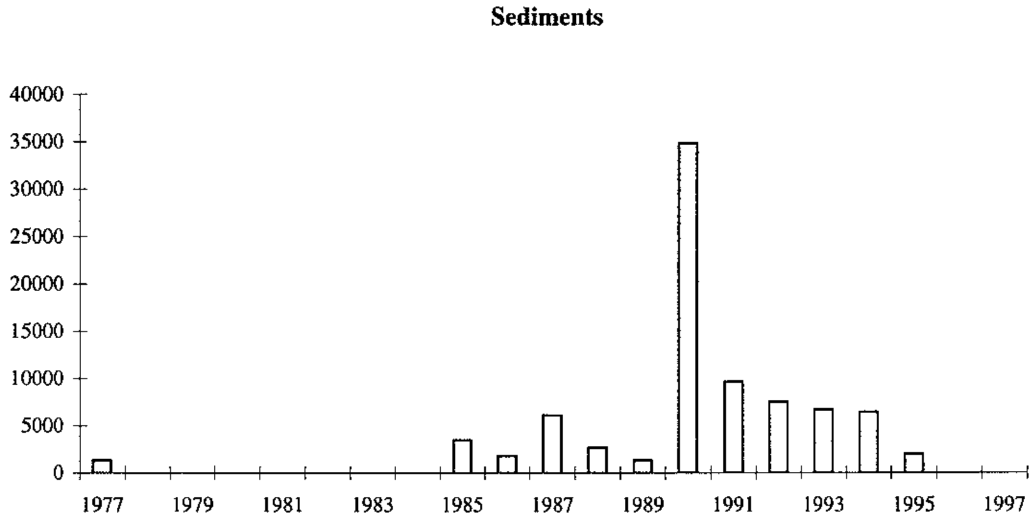
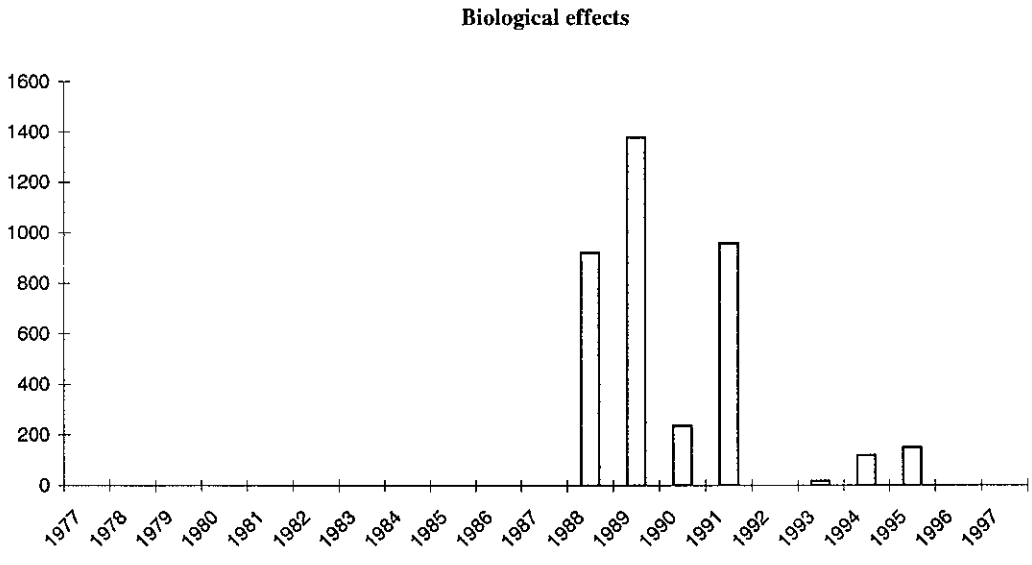



**Figure 19.1.4.** Number of records of data on biological effects of contaminants in the ICES Environmental Data Centre according to year of sampling.



Data generated under the QUASIMEME I exercise are confidential, and no solution has been found for the direct access to these data. However, individual laboratories are free to submit these data to ICES as a voluntary contribution. For data generated under QUASIMEME II, a mechanism has been established that allows individual laboratories to report their data to the ICES Environmental Data Centre. Thus far, no data have been reported by this mechanism.

### 19.1.3 Handling of data for the Arctic Monitoring and Assessment Programme (AMAP)

ICES has continued to serve as the Thematic Data Centre for the marine component of AMAP under a 1998 contact that has been prolonged to cover 1999 as well.

### 19.1.4 European Environment Agency (EEA)

The ICES Environmental Data Centre has received and processed requests for the provision of data on contaminants in biota and sediments for a project on coastal indicators being carried out by the European Topic Centre for the Marine and Coastal Environment.

### 19.1.5 Major data products

The ICES Environmental Data Centre provided information for the Marine Chemistry Working Group (MCWG) in March 1999. Apart from data, the information supplied also included plots and univariate statistics on trace elements in cod liver for the MCWG.

### 19.1.6 On-line access to data inventories

Access to dynamic/query inventories of the data in the ICES Environmental Data Bank can be established through an interface on the ICES website [http://www.ices.dk/env]. The interface allows the user to retrieve information about present holdings of contaminants and biological effects data. It should be noted, however, that not all the data in the ICES Environmental Data Bank are visualized by this web interface as some laboratories have not given permission for their data to be included in this inventory.

### 19.1.7 Fish Disease Database

The ICES Environmental Data Centre has provided information for the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO). This information was used to make an overview of the data available that could be used for a holistic analysis of the ICES fish disease data, and a case study was prepared for the German Bight. The report of this exercise is attached as Annex 5.

### 19.1.8 Marine Mammals By-catch Database

No new data were submitted to the database in 1998/1999.

### 19.1.9 ICES Environmental Data Reporting Format

Members of the OSPAR Working Group on Concentrations, Trends and Effects of Substances in the Marine Environment (SIME) as well as other working groups have commented on the ICES Environmental Data Reporting Format. There are different opinions about the use of this reporting format. Some laboratories have developed their own software for the transfer of data from their own database to the ICES Environmental Data Reporting Format, and these laboratories favour that the reporting format be maintained as it is, while other laboratories suggested that it should be possible to submit data by other means, e.g., in spreadsheet form.

The ICES Environmental Data Centre will review potential alternatives for data submission that would meet the needs of institutes that experience difficulties with the present format without increasing the workload on the Data Centre. It is important, however, to emphasize that this development will demand resources for programming and may take several years to realize.

Hopefully, the development of a user-friendly means of data submission will encourage the laboratories to submit more data to the ICES Environmental Data Centre.

*Recommendations*

In reviewing the activities of the Environmental Data Centre during the past year, the ACME stressed the importance of this database in relation to ICES activities, and not simply in relation to providing a service to the Commissions for their monitoring data. In this connection, the ACME encouraged the submission of data other than those associated with monitoring programmes coordinated by OSPAR, HELCOM, and AMAP, to enhance the use of the database in assessing environmental conditions.

## 19.2 Handling of Nutrient Data for the OSPAR Commission

*Request*

Item 5 of the 1999 Work Programme from the OSPAR Commission: to carry out data handling activities relating to:

5.3 the implementation of the Nutrient Monitoring Programme.

*Source of the information presented*

Progress report from the ICES Oceanographic Data Centre and ACME deliberations.

*Status/background information*

Most nutrient data are normally supplied to the ICES Oceanographic Data Centre via voluntary submissions compiled by National Oceanographic Data Centres or by individual scientists and institutes. Currently, only those data supplied by Belgium, the Netherlands, and Germany are submitted using the OSPAR reporting format, although for various reasons Germany usually resubmits its OSPAR data to ICES using the established international exchange procedures. Germany and Belgium also report OSPAR nutrient-collecting cruises using the Cruise Summary Report (ROSCOP) system that makes it possible to track data submissions, and flag the non-appearance of data. In this way, it is possible to identify expected Belgian data for 1998 (see Table 19.2.1).

In the case of Germany, not enough information has been provided to determine whether any received OSPAR JAMP data refer to that included on the Cruise Summary Forms. Hence, ICES has created forms to describe the data received, on the assumption that the provided forms refer to data not yet received.

Table 19.2.2 lists the number of oceanographic profiles including at least one nitrate measurement, broken down by ICES Member Country and year (since 1990). This illustrates clearly that the problem raised at last year's ACME meeting with regard to the lack of submissions of nutrient data remains, with the number of profiles reduced by more than 50–75 % since the early 1990s. Indeed, a similar analysis for the coastal zone alone would reveal an even greater reduction for this area specifically.

Various attempts to stimulate nutrient submissions via National Data Centres, relevant ICES Working Groups, and OSPAR delegations have so far yielded little reaction.

## 19.3 Development of Biological Databases

*Request*

Item 6 of the 1999 Work Programme from the OSPAR Commission: to continue to establish data banks for phytobenthos, zoobenthos, and phytoplankton species.

Contract from HELCOM to serve as Thematic Data Centre for Baltic Monitoring Programme data for a three-year period beginning on 1 July 1998.

**Table 19.2.1.** Cruises in 1998 collecting data on nutrients for OSPAR, with data yet to be delivered. (Source: ICES ROSCOP.)

| Country | Ship Code | Date | Cruise | No of Stations submitted to ICES |
|---|---|---|---|---|
| Belgium | Belgica A962 | 09–20 Mar 1998 | 98/05 - ref 98/05 | 0 |
| Belgium | Belgica A962 | 15–25 Sep 1998 | 98/20 - ref 98/20 | 0 |
| Belgium | Belgica A962 | 03–06 Nov 1998 | 98/25 - ref 98/25 | 0 |
| Belgium | Belgica A962 | 07–10 Dec 1998 | 98/29 - ref 98/29 | 0 |
| Germany | Haithabu | 28–29 Jan 1998 | ICES created form | 5 |
| Germany | Haithabu | 05–18 Feb 1998 | ICES created form | 6 |
| Germany | Haithabu | 12–26 Aug 1998 | ICES created form | 10 |
| Germany | Haithabu | 03–03 Nov 1998 | ICES created form | 1 |
| Germany | Haithabu | 02–14 Dec 1998 | ICES created form | 10 |
| Germany | Helicopter (NLEG) | 09–09 Feb 1998 | ICES created form | 11 |
| Germany | Helicopter (NLEG) | 16–16 Feb 1998 | ICES created form | 2 |
| Germany | Helicopter (NLEG) | 17–17 Nov 1998 | ICES created form | 11 |
| Germany | Helicopter (NLEG) | 23–23 Nov 1998 | ICES created form | 2 |
| Germany | ng | 01 Jan–31 Dec 1998 | Ref 19990109 | ? |
| Germany | ng | 01 Jan–31 Dec 1998 | Ref 19990059 | ? |

**Table 19.2.2.** ICES profile statistics from 1990–1999 by country (stations with at least one observation of nitrate or nitrate + nitrite).

| Country | 1990 | 1991 | 1992 | 1993 | 1994 | 1995 | 1996 | 1997 | 1998 | 1999 | Total |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 06/07 (Germany) | 1 387 | 1 263 | 1 219 | 1 043 | 265 | 251 | 188 | 144 | 0 | 0 | 5 760 |
| 11 (Belgium) | 123 | 124 | 100 | 169 | 129 | 136 | 118 | 201 | 0 | 0 | 1 100 |
| 18 (Canada) | 31 | 49 | 26 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 106 |
| 26 (Denmark) | 623 | 442 | 681 | 431 | 439 | 386 | 171 | 49 | 31 | 0 | 3 253 |
| 29 (Spain) | 0 | 20 | 29 | 35 | 133 | 43 | 2 | 0 | 0 | 0 | 1 209 |
| 31/32 (US) | 7 | 47 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 54 |
| 34 (Finland) | 218 | 151 | 238 | 149 | 263 | 177 | 86 | 107 | 0 | 0 | 1 389 |
| 35 (France) | 0 | 94 | 24 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 118 |
| 45 (Ireland) | 0 | 0 | 75 | 50 | 0 | 0 | 0 | 0 | 0 | 0 | 125 |
| 46 (Iceland) | 264 | 281 | 328 | 242 | 262 | 0 | 30 | 128 | 140 | 0 | 1 657 |
| 58 (Norway) | 2 033 | 669 | 923 | 842 | 992 | 518 | 369 | 500 | 391 | 0 | 7 237 |
| 64 (Netherlands) | 753 | 1 062 | 983 | 979 | 904 | 881 | 499 | 594 | 55 | 0 | 6 710 |
| 67 (Poland) | 469 | 278 | 17 | 54 | 38 | 41 | 140 | 147 | 0 | 0 | 1 184 |
| 68 (Portugal) | 92 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 92 |
| 74 (UK) | 675 | 330 | 823 | 951 | 391 | 217 | 414 | 178 | 58 | 0 | 4 037 |
| 77 (Sweden) | 936 | 769 | 729 | 492 | 639 | 445 | 411 | 503 | 23 | 0 | 6 183 |
| 90/RU (SU/Russia) | 305 | 161 | 99 | 11 | 52 | 0 | 0 | 0 | 0 | 0 | 628 |
| P.R. (UK) | 648 | 686 | 446 | 1 020 | 0 | 183 | 460 | 164 | 0 | 0 | 3 607 |
| Total | 8 564 | 6 426 | 6 740 | 6 468 | 4 507 | 3 278 | 2 888 | 2 715 | 698 | 0 | 44 449 |

*Source of the information presented*

Progress report from the ICES Environmental Data Centre and ACME deliberations.

*Status/background information*

In 1997 OSPAR and HELCOM requested ICES to prepare reporting formats and data-entry/data-screening software for data on phytoplankton, zooplankton, phytobenthos, and zoobenthos. Subsequently, OSPAR and HELCOM requested ICES to develop a database for these biological parameters. The work is coordinated from the ICES Secretariat, but is also based on input from several Working Groups and individual scientists.

In order to obtain a database structure that is flexible and allows adequate search facilities, graphic representation, and report generation, the new biological database will be based on the use of ACCESS.

A data entry program that facilitates the entry of data into the ICES Biological Database is under development. A prototype of this data entry program was demonstrated for the ICES/HELCOM Steering Group on Quality Assurance of Biological Measurements in the Baltic Sea (SGQAB) and the ICES/OSPAR Steering Group on Quality Assurance of Biological Measurements related to Eutrophication Effects (SGQAE) in February 1999. SGQAB and SGQAE reviewed the draft ICES Biological Data Reporting Format and made several suggestions for amendments and additions; these suggestions will be incorporated into the reporting format as far as possible.

This data entry program will be linked to specific code lists, including species lists/species code lists in order to secure the use of the correct nomenclature of the species. In addition to the data entry program, software will be developed to convert data structured according to the old HELCOM Biological Format into the new format.

In the ICES area, several taxonomic checklists on phytoplankton exist. There is a need to harmonize and combine the present lists and also to update taxonomic checklists regularly in the future. The checklists are needed for the ICES database for proper coding of the species.

## 19.4 Development of Reporting Format for Biological Effects Data

*Request*

Item 7 of the 1999 Work Programme from the OSPAR Commission: to expand the ICES environmental data reporting format to include all the reporting parameters required for each of the biological effects techniques adopted by OSPAR where these reporting formats have not already been developed.

*Source of the information presented*

The 1999 report of the Working Group on Biological Effects of Contaminants (WGBEC) and ACME deliberations.

*Status/background information*

In association with the OSPAR request that ICES expand the Environmental Data Reporting Format to permit the submission of data on the general and contaminant-specific techniques of biological effects monitoring in the Joint Assessment and Monitoring Programme (JAMP), WGBEC reviewed the ICES Environmental Data Reporting Format. WGBEC made a number of suggestions for the addition of new fields and codes to the existing format in order to cover the new biomarkers and monitoring methods included in the JAMP and other programmes. These biomarkers include:

1) P4501A1 (EROD);
2) fluorescent bile metabolites;
3) DNA adducts;
4) fish liver histopathology and liver nodules;
5) imposex and intersex as a result of TBT exposure;
6) metallothionein;
7) δ-amino levulinic acid dehydratase (ALA-D) inhibition in blood;
8) whole sediment bioassays;
9) sediment pore-water bioassays;
10) fish reproductive success;
11) lysosomal stability.

Quality control procedures for the above techniques are being developed under the BEQUALM programme (see Section 7.4, above), and the results of this programme are expected to be fed back to ICES for the final development of the reporting format.

For the best use of these data, the ACME stressed the importance of developing systems that link data on contaminant concentrations, biological effects such as biomarkers, and biological endpoints such as fish diseases, in order to be able to examine cause-effect relationships.

## ANNEX 1

## DETAILED METHODS FOR TEMPORAL TREND DETECTION

### PART 1 ASSESSING THE POWER OF TREND TESTS FOR MONOTONIC TRENDS

### 1.1 Introduction

In this annex, a simple model to evaluate the power of trend assessment methods to detect monotonic trends in annual contaminant indices is presented.

In previous work (see, e.g., ICES, 1997, 1999), the power of the tests under three stylized scenarios has been computed. These scenarios are shown in Figure A1.1.1 and correspond to a

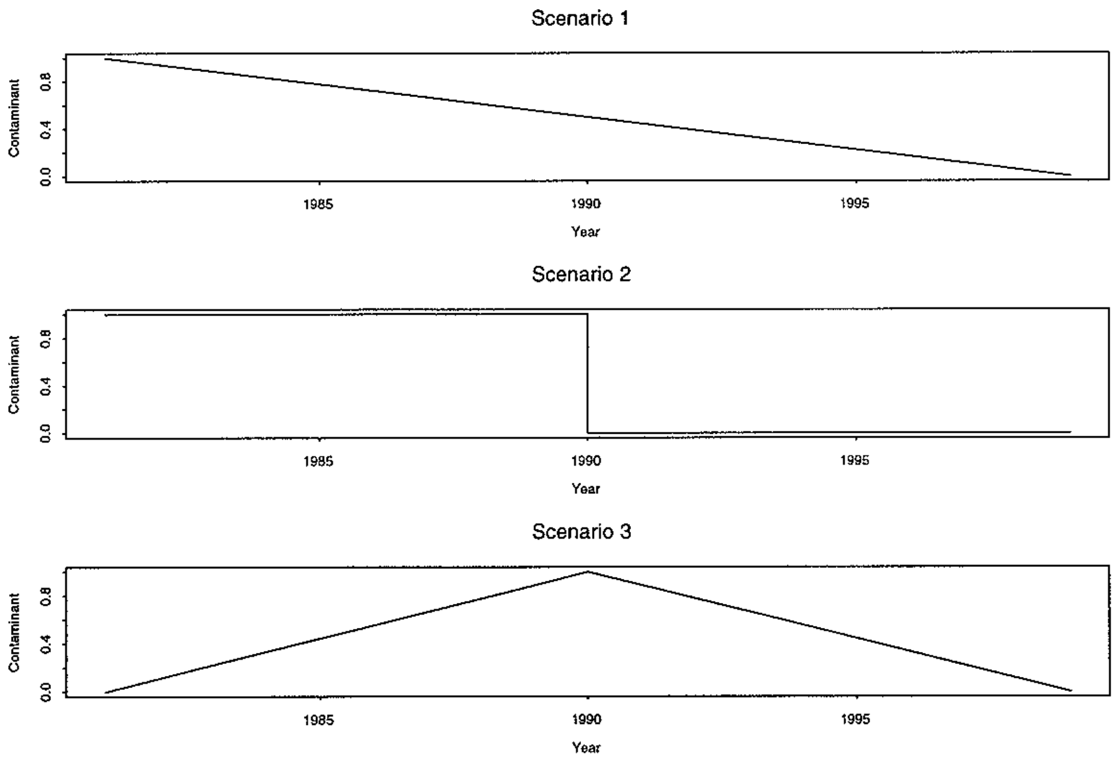
1) linear trend,
2) step trend, and
3) two stage linear trend

respectively.

These scenarios were not necessarily proposed as benchmarks for trend assessment methods. They were selected to represent three different patterns of environmental change that might realistically occur in response to control measures, and were used to demonstrate that the powers of statistical trend tests will depend on the underlying pattern. It has been determined for time series of annual data for 10–20 years that

- methods designed to have good power for detecting monotonic trends (Scenarios 1 and 2) tend to have very poor power for detecting a non-monotonic trend (Scenario 3);
- a general purpose method functioned reasonably well for all three scenarios, but never excelled;
- the trend test based on a seven-year LOESS smoother offered a good compromise—if all three scenarios are of interest (Nicholson *et al.*, 1997).

**Figure A1.1.1.** Three scenarios representing patterns of environmental change that might realistically occur in response to control measures.



### 1.2 A Model for Monotonic Trends

Here we will focus specifically on detecting evidence of a monotonic decline in annual contaminant indices. To compare the power of different trend assessment methods, we will consider a general class of monotonic scenarios.

As a very restricted class, Scenario 1 can be informative. It provides an upper benchmark for methods, and also for evaluating the historic performance of existing monitoring programmes or the design of new ones—if programme performance is poor with Scenario 1, it is likely to be worse when faced with more realistic, less regular trends. However, the power for Scenario 1 provides no indication about the true performance with irregular monotonic trends, and may be overly optimistic when used as a criterion for designing monitoring programmes.

Scenario 2 can also be informative. It is a challenging monotonic trend to detect, but it is still only one of the infinite number of monotonic trends that could flow between a given start and end point.

A more realistic measure of power will be obtained from the average performance over a broad class of scenarios representing a wide range of monotonic trends. The approach we will adopt here is for the $\mu_y$ to follow a constrained random walk between a given start and end point. This is similar to the model fitted to mercury concentrations in cod by Warren (1996).

To construct the random walk, note that the linear trend from $k$ to 0 in $T$ years can be written as

$$\mu_1 = k,$$

$$\mu_y = \mu_{y-1} - \beta \qquad \text{where } \beta = k/(T-1) \text{ for } y = 2 \ldots T.$$

The random walk component increases the variety of monotonic trends by introducing noise into the relationship between $\mu_y$ and $\mu_{y-1}$, with the constraint that $\mu_y \le \mu_{y-1}$ and $\mu_y \ge 0$.

Hence, the model for the annual index $c_y$ is given by

$$c_y = \mu_y + \varepsilon_y \qquad y = 1 \ldots T$$

where

$\varepsilon_y$ is the sampling error in year $y$ with zero mean and standard deviation $\sigma$, and

$$\mu_1 = k$$

$$\begin{aligned} \mu_i &= \mu_{i-1} - \beta + \delta_i && \text{if } 0 \le \mu_{i-1} - \beta + \delta_i \le \mu_{i-1}, \\ &= \mu_{i-1} && \text{if } \mu_{i-1} - \beta + \delta_i > \mu_{i-1}, \text{ or} \\ &= 0 && \text{if } \mu_{i-1} - \beta + \delta_i < 0 \end{aligned}$$

and

$$\mu_T = 0$$

where $\delta_i$ is the stochastic trend component with zero mean and standard deviation $\sigma_R$.

Appropriate values of $\sigma_R$ can be used to generate trend patterns varying from linear ($\sigma_R = 0$) to more complex monotonic trends ($\sigma_R > 0$).

Figure A1.1.2 shows, in the left-hand column, the contaminant time series for single realizations of this model with $T = 15$, $k = 4$, $\sigma = 1$, and $\sigma_R = 0.1, 0.5, 1$, and 2 ($\sigma_R$ is denoted sR in the figures). The solid lines show the constrained random walk in the means, $\mu_y$, and the points correspond to the observed values $c_y$. The right-hand column shows 100 realizations of constrained random walks in the $\mu_y$. The right-hand figures demonstrate the increased spread and complexity of monotonic trends as $\sigma_R$ increases.

### 1.3 Comparison of the Average Power of Mann-Kendall and Smoother Tests with Monotonic Trends

Using computer simulation, we can use the simple model described above to compute the power of different tests averaged over many different monotonic trends. The complexity of these trends can be characterized by the size of the stochastic standard deviation, $\sigma_R$.

For simplicity, we will compare the power of the Mann-Kendall test and the trend test based on a seven-year-window LOESS smoother described in Fryer and Nicholson (1999), currently used in the OSPAR assessments of contaminants in biota. For demonstration, we will assume that the noise components are Normally distributed, which may be appropriate for contaminant concentrations expressed on a log scale.

Figure A1.1.3 shows the power, P, of these two tests as a function of $k/\sigma$ for $T = 10$ and $T = 20$, with $\sigma = 1$, and $\sigma_R = 0, 0.5, 1$, and 2. Each point on each power curve is based on 5000 realizations, giving an accuracy better than 1 % at a power of 90 %.

We see that the initial superiority of the Mann-Kendall test when $\sigma_R = 0$ is progressively eroded as $\sigma_R$ increases beyond $\sigma_R = 0.5$. The test of significance of the smoother reduces only slightly as $\sigma_R$ increases, whereas with $k/\sigma = 4$, the power of the Mann-Kendall test at $\sigma_R = 2$ is considerably reduced, to about half of that at $\sigma_R = 0$.

**Figure A1.1.2.** Contaminant time series for single realizations of the model (left-hand column) and 100 realizations of constrained random walks in the $\mu_y$ (right-hand column).

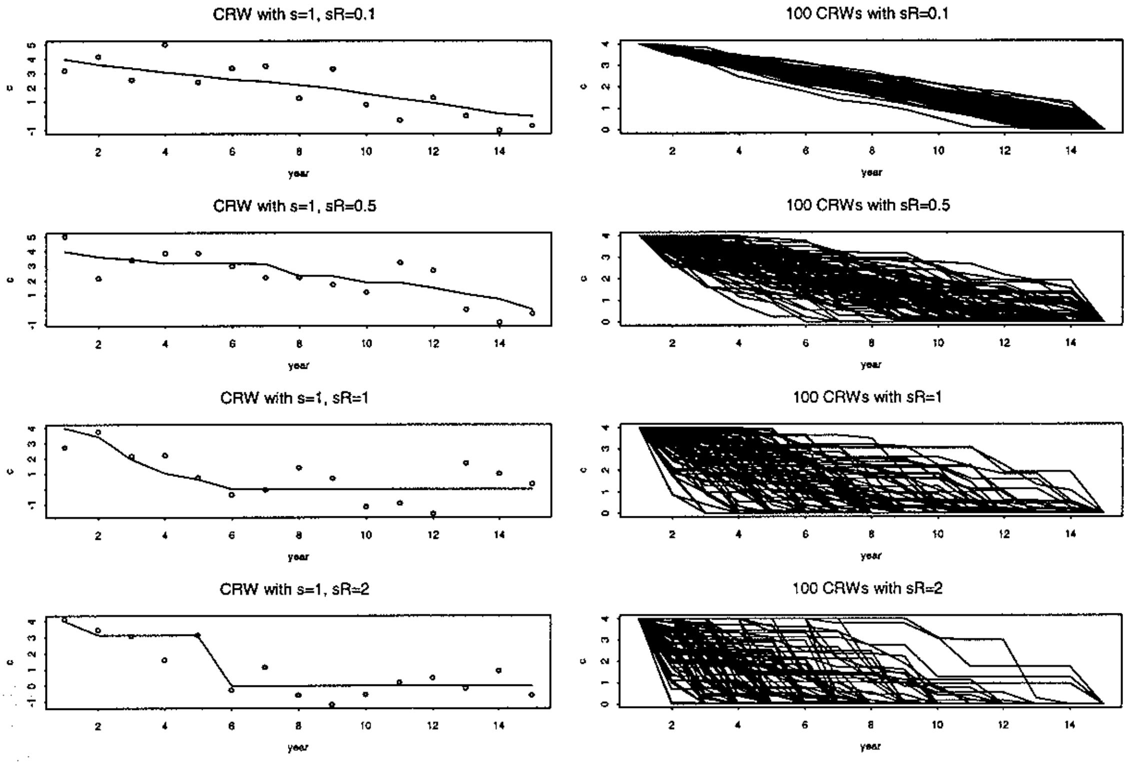
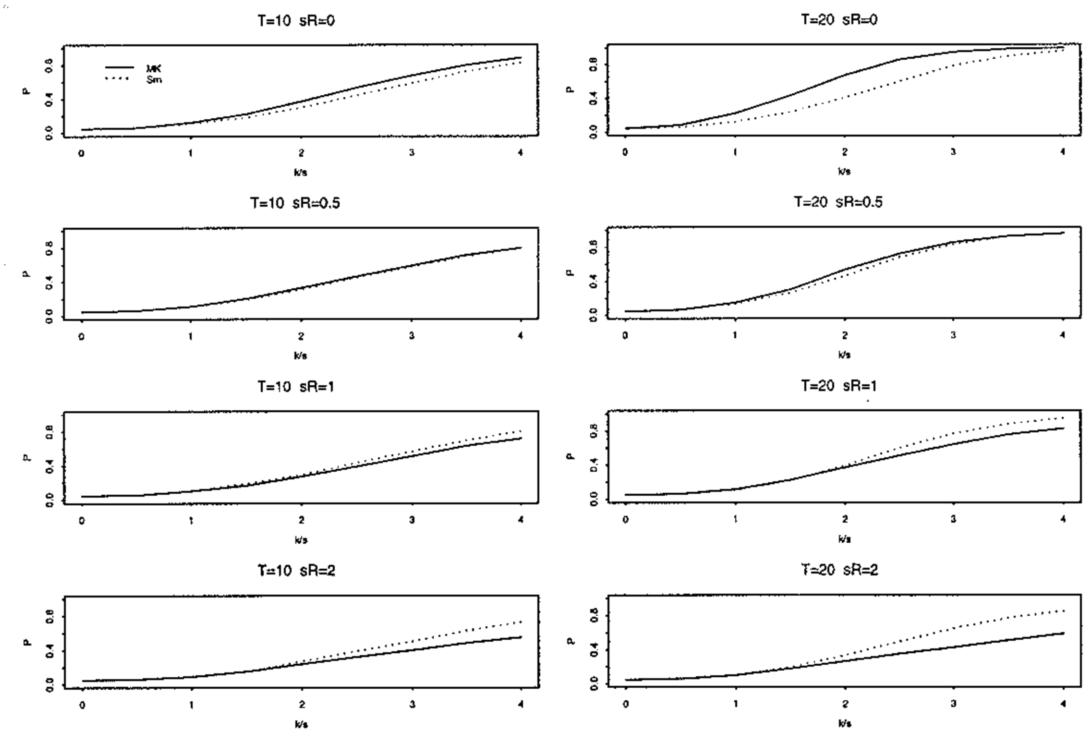



**Figure A1.1.3.** The power of the Mann-Kendall test and the trend test based on the seven-year-window LOESS smoother as a function of $k/\sigma$ for 10 and 20 years.



## 1.4 Discussion and Conclusions

We have presented a simple model for generating complex monotonic trends. The model is a constrained random walk with trend, and the complexity of the trend pattern is characterized by the magnitude of the random component. By averaging over a large number of realizations of this model, a measure of the average power for detecting monotonic trends of a given complexity can be obtained. This approach may be useful for comparing methods, as here, or for providing a more realistic measure of the performance of existing programmes or proposed new ones.

The random-walk-with-trend model is sensible for contaminant time series where a smooth underlying trend is buffeted by reality. However, the constraints we have introduced to induce monotonic trends with a specified trend effect ($k/\sigma$) may give unrealistic trends for large values of $\sigma_R$. As this increases, the distribution of trends above and below a linear trend becomes progressively more asymmetric, with a greater proportion below. In the extreme, only values of $k$ or 0 will tend to be observed, with a 50 % chance of indices with value $k$ becoming 0 at each step. Further, since $\mu_{T-1}$ must drop to 0 whatever its value, the decrease from $\mu_{T-1}$ to 0 has different properties from the decrements between other years.

A more elegant solution might be to simulate from a Dirichelet distribution with parameters $\alpha_1$, $\alpha_2$,..,$\alpha_{T-1}$. This would generate $T-1$ random deviates that sum to unity and, hence, constrained transitions in $T-1$ steps from 0 to 1. Taking $\alpha=\alpha_1=\ldots\alpha_{T-1}$ would ensure that the mean of each transition was $1/(T-1)$. Small values of $\alpha$ would give complex monotonic changes. Large values of $\alpha$ would give smooth, almost linear, changes.

For the values of $\sigma_R$ used here, the model behaves reasonably well.

The results of the simulations showed that, as would be expected, the Mann-Kendall test is superior to the smoother where there is a near-linear monotonic trend. This is reversed in favour of the smoother as monotonic trends become progressively more complex. However, the robustness and simplicity of the Mann-Kendall test may be a consideration in a particular application.

If the objective is only to detect monotonic trends, the sharper focus of the Mann-Kendall test could also be important. Although the smoother is better at detecting complex monotonic changes, it will also detect troublesome non-monotonic changes, unless some way of separating them is incorporated.

## 1.5 Acknowledgement

This material has been prepared by R.J. Fryer, FRS Marine Laboratory, Aberdeen, UK, and M.D. Nicholson, CEFAS, Lowestoft Laboratory, UK.

## 1.6 References

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ICES. 1997. Report of the ICES Advisory Committee on the Marine Environment, 1997. ICES Cooperative Research Report, 222: 26–29.

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Nicholson, M.D., Fryer, R.J., and Maxwell, D.L. 1997. A study of the power of various methods for detecting trends. ICES CM 1997/Env:11.

Warren, W.G. 1996. Kalman filter for a simple time series: mercury concentrations in cod muscle. ICES CM 1996/D:1.

# PART 2 COMPOUND MANN-KENDALL TEST

## 2.1 Summary

Two complementary non-parametric tests of trend are combined to provide a single test suitable for detecting a broad range of trend patterns. Appropriate critical values for each component test are derived such that the size of the combined test is controlled at, e.g., 5 %. The power of the combined test is compared with other methods for testing broad patterns of temporal change.

## 2.2 Introduction

The Mann-Kendall test (Mann, 1945; Kendall, 1975) is a commonly used non-parametric test for detecting temporal trends. It is robust and powerful for monotonic trends, but has poor power to detect non-monotonic trends (Nicholson *et al.*, 1997, 1998). The non-linear Mann-Kendall test (El-Shaarawi and Niculescu, 1993) is an extension of the Mann-Kendall test to detect non-linear trends, however, this has no power to detect linear trends.

This paper describes and studies the compound Mann-Kendall (CMK) test, a combination of the Mann-Kendall and non-linear Mann-Kendall tests, as a general non-parametric method for detecting trends.

Section 2.3 describes the CMK test. Section 2.4 shows how the critical values for the test were calculated. Section 2.5 is a power comparison, taken from Fryer and Nicholson (1999), for the CMK test, a locally weighted regression smoother (Cleveland, 1979, 1993), and the Successive Differences test (von Neumann, 1941).

### 2.3 Description of the Compound Mann-Kendall Test

The CMK test consists of calculating both the Mann-Kendall test statistic $S$, and the non-linear Mann-Kendall test statistic $S_\beta$. If either is significant, the CMK test is judged to be significant.

The Mann-Kendall test statistic $S$ is generated by comparing the size of each observation with that of each observation that follows it. So, for a series of annual indices $y_t$, observed in year $t = 1,\ldots,T$:

$$S = \sum_{k=1}^{T-1} \sum_{j=k+1}^{T} \operatorname{sgn}(y_j - y_k)$$

where

$$\begin{aligned} \operatorname{sgn}(y_j - y_k) &= 1 \quad \text{if } y_j - y_k > 0 \\ &= 0 \quad \text{if } y_j - y_k = 0 \\ &= -1 \quad \text{if } y_j - y_k < 0. \end{aligned}$$

For the non-linear Mann-Kendall test statistic, the successive differences between the observations $y_t$ are calculated to remove any linear trend and the Mann-Kendall test statistic is calculated for these differences, i.e.,

$$\beta_t = y_{t+1} - y_t$$

$$S_\beta = \sum_{k=1}^{T-2} \sum_{j=k+1}^{T-1} \operatorname{sgn}(\beta_j - \beta_k)$$

Details of significance testing for the individual tests are reproduced from the relevant references in the Appendix. However, for the CMK test, empirical critical values were calculated to ensure that the test had the required size.

### 2.4 Setting Critical Values for the CMK Test

The CMK test is based on the joint distribution of $S$, the Mann-Kendall test statistic, and $S_\beta$, the test statistic from the non-linear Mann-Kendall test. The empirical joint distribution of $S$ and $S_\beta$, under the null hypothesis of no trend, was used to select the critical values for the CMK test and to look at the association between the two statistics.

For $T = 10$ and $T = 20$ years, 100 000 series of $T$ independent, N(0,1) distributed observations were generated. $S$ and $S_\beta$ were calculated for each series and the empirical joint distribution of the two statistics was recorded. From this, the values in Tables A1.2.1 and A1.2.2 were calculated.

Table A1.2.1 ($T = 10$) and Table A1.2.2 ($T = 20$) show:

1) the proportion of the 100 000 series with $|S|$ **OR** $|S_\beta|$ greater than the values stated in the row and column headings. These are the bold figures in the body of the table.

2) the proportion of the series with the statistic greater than the stated value for each test separately. These are the figures in the second row and second column.

3) the expected proportion with $|S|$ **OR** $|S_\beta|$ greater than the stated values calculated from 2) assuming that the results for each test statistic are independent. These are the smaller figures in the body of the table.

Note that $S$ and $S_\beta$ are discrete by definition and they increase in steps of size 2 as no ties are present in the generated data.

For significance testing at the 5 % level, the aim was to select a pair of values for $S$ and $S_\beta$ so that the CMK test had size 5 % and the Mann-Kendall and non-linear Mann-Kendall components had equal weight.

For $T = 20$, Table A1.2.2 shows that using $|S| > 68$ OR $|S_\beta| > 29$ as the critical value gives the CMK test the desired size and the two components have similar weight.

For $T = 10$, Table A1.2.1 shows that none of the combinations give the CMK test the desired properties. When the size is 0.04933, the contribution of the two components is not equal.

Therefore, the required size of the CMK test for $T=10$ was achieved by including a random element in the significance criteria, as follows:

if $|S| > 25$ OR $|S_\beta| > 14$ result judged significant,

if $|S| = 25$ generate $u_1 \sim U(0,1)$, if $u_1 < \rho_1$ result judged significant,

if $|S_\beta| = 14$ generate $u_2 \sim U(0,1)$, if $u_2 < \rho_2$ result judged significant.

The inclusion of this random element in the test sets the power between that of using

$|S| > 23$ OR $|S_\beta| > 12$ and that of using

$|S| > 25$ OR $|S_\beta| > 14$.

$\rho_1$ and $\rho_2$ were calculated as:

required probability of judging result significant = 0.05

1−(1−probability that $|S|$ judged significant) × (1−probability that $|S_\beta|$ judged significant) = 0.05

**Table A1.2.1.**

| $T = 10$ prop. $\lvert S\rvert > X$ OR $\lvert S_\beta\rvert > Y$ | Y | 10 | 12 | 14 |
|---|---|---|---|---|
| | proportion $\lvert S_\beta\rvert > Y$ | 0.06892 | 0.03303 | 0.01401 |
| X | proportion $\lvert S\rvert > X$ | | | |
| 21 | 0.04572 | **0.11193** 0.11149 | **0.07754** 0.07724 | **0.05928** 0.05909 |
| 23 | 0.02847 | **0.09568** 0.09543 | **0.06077** 0.06056 | **0.04222** 0.04208 |
| 25 | 0.01680 | **0.08461** 0.08456 | **0.04933** 0.04928 | **0.03063** 0.03057 |

**Table A1.2.2.**

| $T = 20$ prop. $\lvert S\rvert > X$ OR $\lvert S_\beta\rvert > Y$ | Y | 27 | 29 | 31 |
|---|---|---|---|---|
| | proportion $\lvert S_\beta\rvert > Y$ | 0.03868 | 0.02772 | 0.01932 |
| X | proportion $\lvert S\rvert > X$ | | | |
| 66 | 0.02807 | **0.06576** 0.06566 | **0.05508** 0.05501 | **0.04687** 0.04685 |
| 68 | 0.02325 | **0.06114** 0.06103 | **0.05037** 0.05033 | **0.04211** 0.04212 |
| 70 | 0.01925 | **0.05725** 0.05719 | **0.04645** 0.04644 | **0.03818** 0.03820 |

Assuming that the test statistics are independent and equal weight is required gives:

probability that $\lvert S\rvert$ judged significant = probability that $\lvert S_\beta\rvert$ judged significant = 0.02532

so,

prob($\lvert S\rvert > 25$) + $\rho_1$ prob($\lvert S\rvert = 25$) = 0.02532,
$\therefore \rho_1 = 0.73008$

prob($\lvert S_\beta\rvert > 14$) + $\rho_2$ prob($\lvert S_\beta\rvert = 14$) = 0.02532,
$\therefore \rho_2 = 0.59464$

The assumption that the test statistics are independent appears adequate, comparing the observed proportions to those expected under the assumption of independence (see Table A1.2.1). This assumption was backed up when the power comparison described in Section 2.5 was run. For the 15 000 simulation runs with $T = 10$ and zero trend, the mean power of the CMK test was 0.00499.

Note that obtaining a required size for the CMK test by selecting a proportion of cases for certain values of the test statistics may not generally be necessary. It was done here to ensure a fair comparison of the methods' power in Section 2.5.

In practice, using $\lvert S\rvert > 23$ OR $\lvert S_\beta\rvert > 12$ as a slightly more liberal test criterion or using $\lvert S\rvert > 25$ OR $\lvert S_\beta\rvert > 14$ as a slightly more conservative test criterion maybe adequate.

## 2.5 Power Comparison

The power of the CMK test and two other portmanteau tests of temporal trends are assessed in the following power comparison, taken from Fryer and Nicholson (1999).

The two other tests are:

- *Smoother*—specifically, a locally weighted linear regression smoother with a fixed window width of seven years.
- *Successive Differences test*—a non-parametric test based on the mean squared difference between successive observations. Significance was assessed using the critical values tabulated by Bissell and Williamson (1988).

The power depends upon the underlying pattern of change, the signal-to-noise ratio of the trend, the length of the series, and the significance level of test. The power is considered for three patterns of change in $E[y_t]$, the expected value of the contaminant level at time $t$.

These three patterns are shown as Scenarios 1, 2, and 3 in Figure A1.2.1. They represent a simple, basic trend, a difficult monotonic trend, and a difficult non-monotonic trend, respectively.

The signal-to-noise ratio is quantified here as $k/\psi$, where $k$ is the difference between the smallest and largest value of $E[y_t]$ and $\psi$ is the between-year standard deviation of gaussian noise superimposed on each scenario.

Series of length $T = 10$ and $T = 20$ years are considered and a significance level of 5 % is used for the tests.

Power curves were computed for each test by simulating the test 5000 times for a range of values of $k/\psi$. This gives power curves estimated, with 95 % confidence, to within ± 1 % in the region of 90 % power.

The power curves are shown in Figure A1.2.2. The CMK test has good power for Scenario 1 and for Scenario 2 with $T = 20$. It has the lowest power of the three tests for Scenario 3 and for Scenario 2 with $T = 10$. This is due to the high power of the Mann-Kendall test for detecting monotonic trends and the relatively low power of the non-linear Mann-Kendall test for detecting the non-monotonic trend in Scenario 3.

## 2.6 Conclusions

The CMK test combines some ability to detect non-linear trends with the benefits of the Mann-Kendall test, i.e., it is non-parametric and has good power to detect monotonic trends. It may be suitable to use in place of the Mann-Kendall test when detecting monotonic trends would be of main importance but strong non-linear trends would also give cause for concern. If only monotonic trends are of concern, the Mann-Kendall test is preferable.

For a general purpose test to detect trends, the smoother is preferable to the CMK test. The smoother has relatively good power for all three scenarios. As explained in Fryer and Nicholson (1999), the smoother can also be used to make predictions of the future contaminant levels and has the advantage of providing a useful graphical summary of the trend.

## 2.7 Acknowledgement

This material was prepared by D. Maxwell, CEFAS, Lowestoft Laboratory, UK.

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**Figure A1.2.1.** The three stylized patterns of change in contaminant levels used for the power comparison.

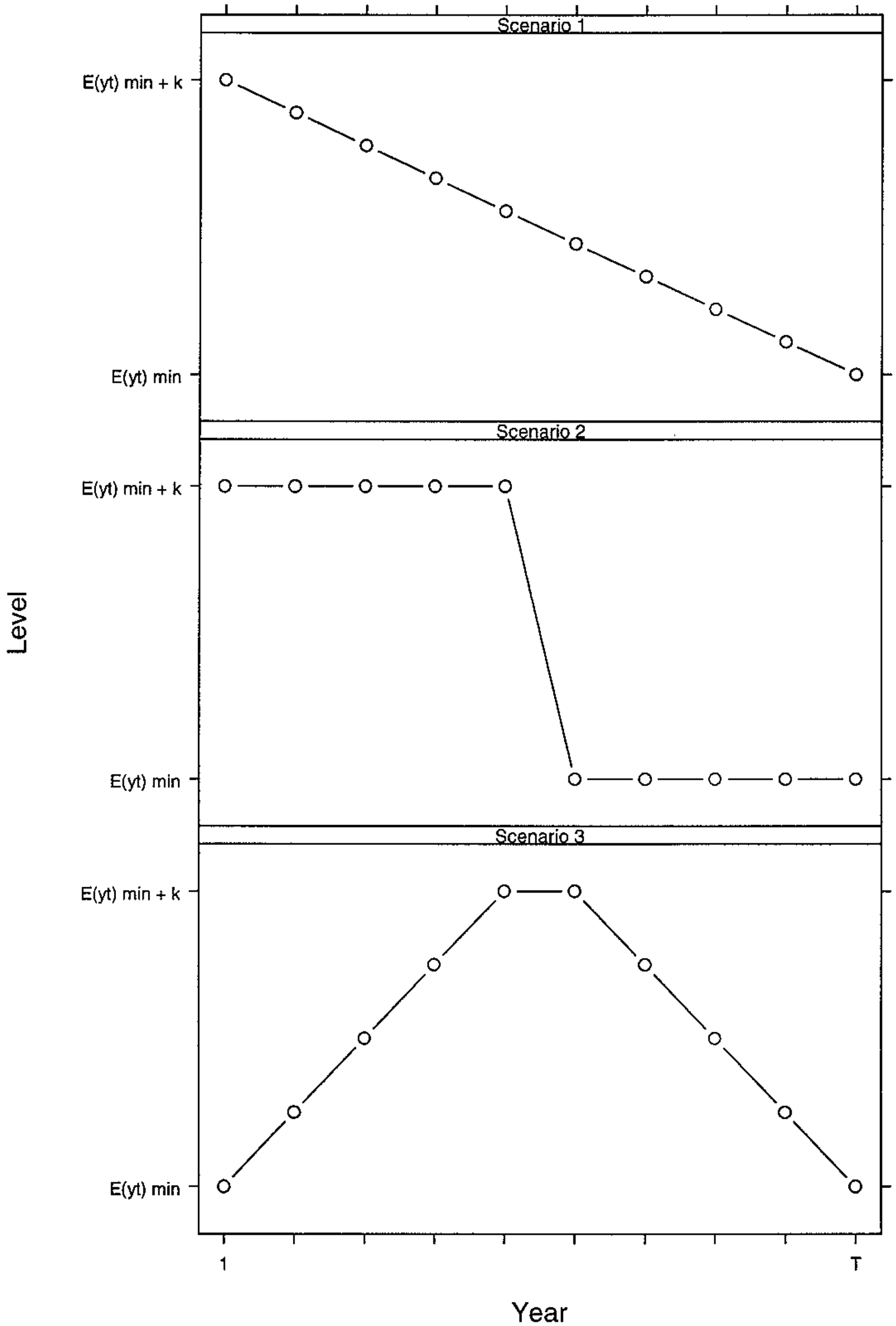
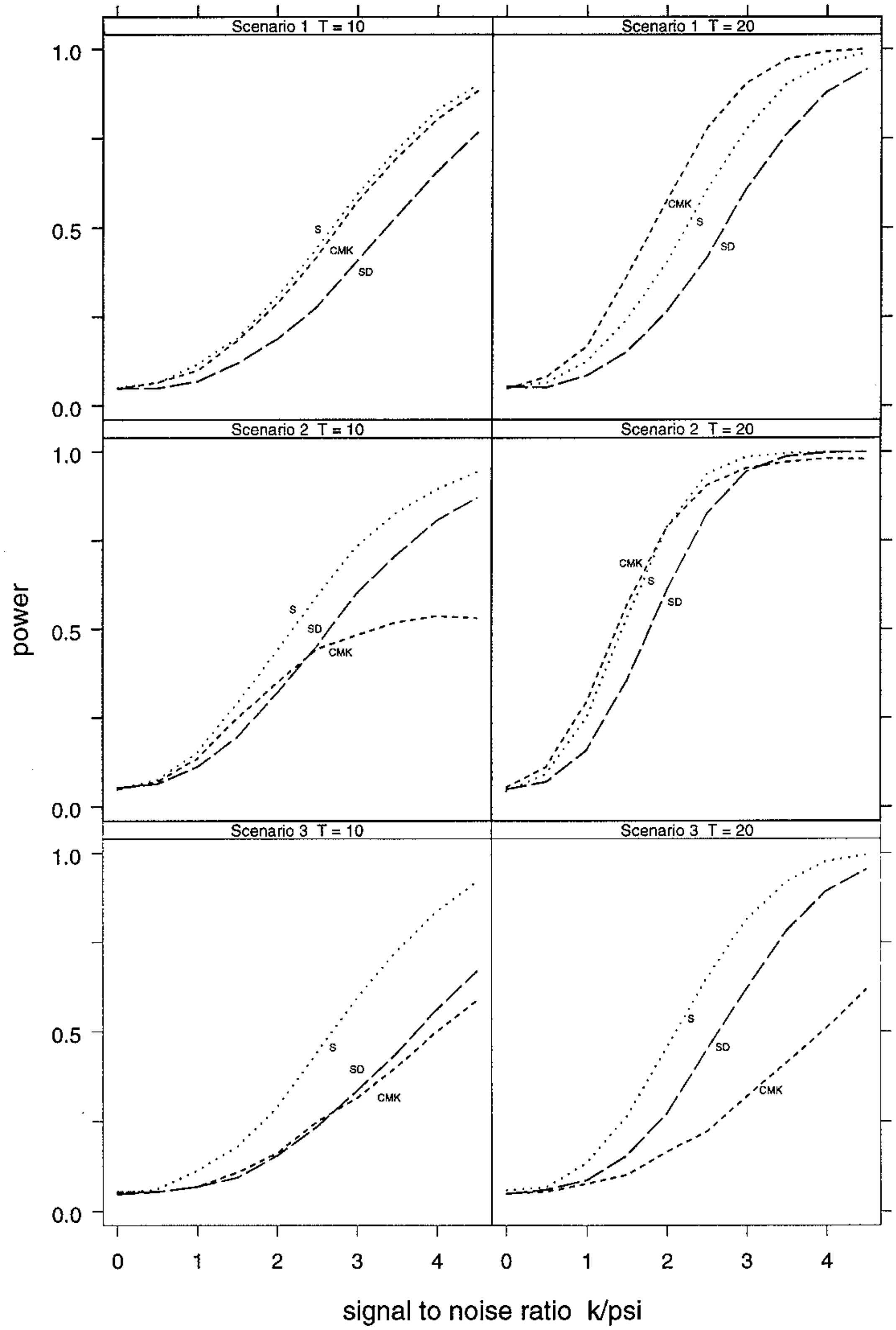


**Figure A1.2.2.** Power curves for tests of temporal trends: CMK = compound Mann-Kendall test; S = smoother; SD = successive differences test.



## Appendix 1

### Significance Testing for the Mann-Kendall and Non-Linear Mann-Kendall Tests

**Mann-Kendall test**

Probability values for small values of $T$ are tabulated in Kendall (1975) and reproduced in, e.g., Gilbert (1987). Alternatively, the variance of $S$ is given which leads to an asymptotically Normal test.

$$\mathrm{var}(S) = \frac{1}{18}\left[T(T-1)(2T+5) - \sum_{p=1}^{g} t_p(t_p-1)(2t_p+5)\right]$$

where $g$ is the number of tied groups and $t_p$ the number of observations in the $p$'th tied group, then the test statistic is

$$Z = \begin{cases} (S-1)(\mathrm{var}(S))^{-1/2} & \text{if } S > 0 \\ 0 & \text{if } S = 0 \\ (S+1)(\mathrm{var}(S))^{-1/2} & \text{if } S < 0 \end{cases}$$

which is compared to the quantiles from a standard Normal distribution.

**Non-linear Mann-Kendall test**

For $T \geq 8$, El-Shaarawi and Niculescu (1992) showed that:

$$var\,(S_\beta) = \frac{1}{18}(2T^3 - 15T^2 + 67T - 102) + 4(T-3)K_1 + 2(T-4)(T-3)K_2$$
$$+ 2(T-4)(K_3 + K_6 + K_8) + 2(T-5)(T-4)(K_4 + K_9) + \frac{2}{3}(T-6)(T-5)(T-4)K_5 + 2(T-3)K_7$$

where assuming Normal errors for the data:

| $K_1$ | $K_2$ | $K_3$ | $K_4$ | $K_5$ | $K_6$ | $K_7$ | $K_8$ | $K_9$ |
|---|---|---|---|---|---|---|---|---|
| 0.2677 | 0.1609 | –0.1609 | –0.3333 | –0.1609 | –0.2677 | –0.4646 | 0.1066 | 0.0000 |

Let,

$$C_\beta = S_\beta \text{ var } (S_\beta)^{-1/2}$$

Then under the null hypothesis $H_0^L$: the trend is linear, as $T \rightarrow \infty$, the asymptotic distribution of $C_\beta$ is standard Normal. They consider that the distribution of $C_\beta$ is well approximated by the standard Normal distribution for $T \geq 20$.

**References**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## PART 3 ASSESSING THE ACCURACY OF ESTIMATORS FOR THE PERCENTAGE OF REDUCTION OF THE CONTAMINANT LEVEL

### 3.1 Introduction

In order to assess environmental changes that might occur in response to control measures, a statistical analysis based on a time series of annual contaminant indices (such as annual riverine or atmospheric inputs) should comprise:

- a statistical trend test and/or confidence limits for the estimated trend,
- an estimate of the underlying smooth trend,
- an estimate of the percentage of reduction of the contaminant level.

The trend test answers the question of whether there is a trend. If there is a trend, an important aspect of the trend is the estimated percentage of reduction of the contaminant level in the observed period of time $[1,T]$. To be consistent, it should be calculated from the estimated trend function. In many cases, the statistical analysis is based on the log concentration. If $\mu_t$ denotes the estimated trend function of the logarithm of the contaminant level in year $t = 1,...,T$, the percentage of reduction of the level in the time period $[1,T]$ may be expressed as

$$100\,(1-\mathrm{EXP}(\mu_T)/\mathrm{EXP}(\mu_1)) = 100\,(1-\mathrm{EXP}(\mu_T-\mu_1))$$

$$= 100\,(1-\mathrm{EXP}((T-1)\beta)),$$

where $\beta$ denotes the average decline of the trend between year 1 and year $T$.

In this paper, two techniques to estimate the percentage of reduction of the contaminant level are compared with regard to their estimation bias and standard deviation. The comparison is based on the model for monotonic non-linear trends proposed by Nicholson and Fryer, as described in Part 1 of this Annex.

### 3.2 Model for Monotonic Trend

According to the model of Nicholson and Fryer (this Annex, Part 1), the logarithm $c_t = \ln(L_t)$ of the annual contaminant level $L_t$ in year $t = 1,...,T$ may be expressed as

$$c_t = \mu_t + \varepsilon_t$$

where

$\varepsilon_t$ is the random deviation in year $t$ with zero mean and standard deviation $\sigma$,

$\mu_1 = k > 0,$
$\mu_T = 0,$

$$\mu_t = \begin{cases} \nu_t & if \quad 0 \le \nu_t \le \mu_{t-1} \\ \mu_t & if \quad \nu_t > \mu_{t-1} \\ 0 & if \quad \nu_t < 0 \end{cases} \qquad \text{for } 1 < t < T, \text{ where}$$

$$\nu_t = \mu_{t-1} - \beta + \delta_t,$$

$\beta = k/(T-1)$, and

$\delta_t$ = stochastic trend component with zero mean and standard deviation $\sigma_R$.

All random components of this model are assumed to be stochastically independent.

Note that in this model the start point and end point of the trend course are fixed, but in between these points, the course is influenced by a random walk. The percentage of reduction of the level in the time period $[1,T]$ may be calculated as

$$100\,(1-\mathrm{EXP}(\mu_T-\mu_1)) = 100\,(1-\mathrm{EXP}(-k)).$$

For further details of the model, the reader is referred to Part 1 of this Annex.

### 3.3 Comparison of Estimators for the Percentage of Reduction Derived from Theil Slope and LOESS Smoother under Normal Distribution

The percentage of reduction of the contaminant level may be estimated by

$$100\,(1-\mathrm{EXP}((T-1)b)$$

where $b$ denotes the Theil slope applied to the logs $c_t = \ln(L_t)b$. It may be estimated alternatively using the seven-year-window LOESS smoother described in Fryer and Nicholson (1999).

The averaged bias and standard deviation of these estimators were computed with computer simulation (5000 runs) under the model described in the preceding section. The model allows different specifications of the underlying distributions. The results presented in this section were computed under the assumption that both the stochastic trend component $\delta_t$ and the noise component $\varepsilon_t$ are normally distributed, i.e.,

$\delta_t \sim N(0,\sigma_R^2)$ and $\varepsilon_t \sim N(0,\sigma^2)$.

Table A1.3.1 contains the results for $T = 10$ and several settings of the noise standard deviation and the standard deviation of the stochastic trend component variance components. For each setting, the results are summarized in a 2 x 2 matrix: the first row contains the mean and the second row the estimation standard deviation for the LOESS smoother (first column) and the Theil slope (second column), respectively. The reduction rate is assumed to be 50 %, which corresponds to $k = -\ln(0.5) = 0.693$.

It turns out that under all settings the estimator based on the LOESS smoother outperforms the Theil slope estimator with regard to both the estimation standard deviation and the bias. The differences are small if the stochastic trend component is small relative to the noise component ($\sigma_R << \sigma$), but the advantages of the LOESS smoother become substantial if $\sigma_R$ and $\sigma$ are of the same order of magnitude. A rather atypical behaviour can be observed at $\sigma_R = 0.2$: the accuracy of the estimator derived from the Theil slopes increases as $\sigma$ increases from $\sigma = 0.1$ to $\sigma = 0.3$. This is caused by the large percentage of vanishing $\mu_t$'s if $\sigma_R = 0.2$: the Theil slope estimator does not function satisfactorily if there is a decline of the level in a first period and a constant level in a second period.

Table A1.3.2 contains the results for $T = 20$. Again, under all settings the estimator based on the LOESS smoother outperforms the Theil slope estimator. The differences are even more substantial than in the case $T = 10$. A rather atypical behaviour can be observed if the results for $T = 10$ and $T = 20$ are compared: for $T = 10$ the Theil slope appears to be better than for $T = 20$. This is a specific property of the Theil slope estimator: it gets worse if, after a period of decline, another period with constant level is appended.

**Table A1.3.1.** Averaged mean and standard deviation for the estimators of the percentage of reduction derived from LOESS smoother and Theil slope. $T = 10$. $\delta_t \sim N(0,\sigma_R^2)$, $\varepsilon_t \sim N(0,\sigma^2)$. True percentage of reduction: 50 %.

| **Mean** / **s.d.** | **$\sigma = 0.1$** | | **$\sigma = 0.3$** | |
|---|---|---|---|---|
| | LOESS | THEIL | LOESS | THEIL |
| **$\sigma_R = 0$** | 49.99 | 49.99 | 49.72 | 49.71 |
| | 1.552 | 1.598 | 4.602 | 4.818 |
| **$\sigma_R = 0.03$** | 49.8 | 49.65 | 49.64 | 49.55 |
| | 2.467 | 3.601 | 5.133 | 5.729 |
| **$\sigma_R = 0.1$** | 49.64 | 48.24 | 49.34 | 48.81 |
| | 4.097 | 7.283 | 6.084 | 7.764 |
| **$\sigma_R = 0.2$** | 49.31 | 43.86 | 49.09 | 46.53 |
| | 5.202 | 12.41 | 7.032 | 11.27 |

**Table A1.3.2.** Averaged mean and standard deviation for the estimators of the percentage of reduction derived from LOESS smoother and Theil slope. $T = 20$. $\delta_t \sim N(0,\sigma_R^2)$, $\varepsilon_t \sim N(0,\sigma^2)$. True percentage of reduction: 50 %.

| **Mean** / **s.d.** | **$\sigma = 0.1$** | | $\sigma = 0.3$ | |
|---|---|---|---|---|
| | LOESS | THEIL | LOESS | THEIL |
| **$\sigma_R = 0$** | 49.95 | 49.96 | 49.85 | 49.94 |
| | 1.527 | 1.144 | 4.73 | 3.593 |
| **$\sigma_R = 0.03$** | 49.52 | 49.82 | 49.44 | 49.84 |
| | 2.327 | 4.626 | 4.978 | 5.361 |
| **$\sigma_R = 0.1$** | 49.92 | 47.62 | 49.82 | 49.12 |
| | 2.574 | 9.135 | 5.142 | 8.271 |
| **$\sigma_R = 0.2$** | 49.47 | 34.76 | 49.4 | 40.7 |
| | 3.92 | 16.81 | 5.983 | 14.22 |

In conclusion, under Normal distribution there is no reason to use the Theil slope instead of the LOESS smoother. Differences are small as long as the stochastic trend component is negligible, but it will become substantial when the monotonic trend is clearly non-linear. For short time series ($T = 10$) and large noise standard deviation ($\sigma = 0.3$), the advantages of the LOESS smoother relative to the Theil slope are not as large as for the other settings.

### 3.4 Comparison of Estimators for the Percentage of Reduction Derived from Theil Slope and LOESS Smoother under Outlier-Contaminated Normal Distribution

This section reports the outcome of a computer simulation based on the same model as in the preceding section. The only difference concerns the distribution of the noise component $\varepsilon_t$, which is now a mixture of two Normal distributions with zero mean but different variances:

$$0.9\ N(0,p^2) + 0.1\ N(0,100\,p^2),$$

i.e., with 90 % probability that the standard deviation equals $p$, and with 10 % probability that the standard deviation equals $10p$. This distribution can be interpreted as a mixture of a regular distribution and an outlier distribution, but it can also be interpreted as a temporal heteroscedastic model. The variance of the mixture of both distributions can be calculated as:

$$\mathrm{Var}(\varepsilon_t) = 0.9\,p^2 + 0.1 \times 100\,p^2 = 10.9\,p^2.$$

$p$ is chosen so that $\mathrm{Var}(\varepsilon_t)$ remains being $\sigma^2$, i.e., $p = \sigma/\sqrt{10.9}$. The stochastic trend component and all other settings are specified as in the preceding section, i.e., $\delta_t \sim N(0,\sigma_R^2)$.

Table A1.3.3 contains the results for $T = 10$. We see that, in the presence of outliers, the performance of LOESS is poorer, with an increased variance although still approximately unbiased. The performance is relatively unaffected by increasing $\sigma_R = 0.2$. In comparison, the Theil estimator appears to be unaffected by outliers. Its performance is superior to that of LOESS for small values of $\sigma_R$ (0 and 0.03), being approximately unbiased with smaller variance. However, for larger values of $\sigma_R$, the Theil estimator becomes more biased with an increasingly larger variance.

With $\sigma = 0.3$, the performance of the LOESS estimator is considerably worse. It is again relatively unaffected by the size of $\sigma_R$, but the presence of outliers introduces some bias and an inflated variance. The Theil estimator is a non-parametric method and, therefore, less affected. It is again unbiased for small $\sigma_R$, with both increasing bias and variance as $\sigma_R$ increases. However, the expected error of the Theil estimator is generally considerably lower than that of the LOESS estimator. This is due to the outlier sensitivity of the LOESS estimator.

Table A1.3.4 contains the results for $T = 20$. Again, in comparison with Table A1.3.3, the effect of outliers on the LOESS estimator is to increase the variance but not the bias; the performance of the Theil estimator is a slight increase in variance. The Theil estimator outperforms the LOESS estimator for small values of $\sigma_R$. However, with $\sigma = 0.1$, the Theil estimator becomes more biased, with a larger variance with increasing $\sigma_R$. The changes in performance are similar with $\sigma = 0.3$, although now the performance of the Theil estimator is superior except for $\sigma_R = 0.2$, where, although the variance is smaller, the bias is larger.

**Table A1.3.3.** Averaged mean and standard deviation for the estimators of the percentage of reduction derived from LOESS smoother and Theil slope. $T = 10$. $\delta_t \sim N(0,\sigma_R^2)$, $\varepsilon_t$ outlier-contaminated. True percentage of reduction: 50 %.

| **Mean** / **s.d.** | $\sigma = 0.1$ | | $\sigma = 0.3$ | |
|---|---|---|---|---|
| | LOESS | THEIL | LOESS | THEIL |
| $\sigma_R = 0$ | 49.74 | 49.95 | 47.16 | 49.51 |
| | 5.279 | 2.211 | 20.92 | 7.566 |
| $\sigma_R = 0.03$ | 49.55 | 49.59 | 47.32 | 49.19 |
| | 5.746 | 4.174 | 19.54 | 7.948 |
| $\sigma_R = 0.1$ | 49.54 | 48.45 | 47 | 48.41 |
| | 6.461 | 7.904 | 18.96 | 10.17 |
| $\sigma_R = 0.2$ | 49.1 | 44.76 | 46.98 | 46.15 |
| | 7.272 | 12.34 | 20.37 | 14.73 |

**Table A1.3.4.** Averaged mean and standard deviation for the estimators of the percentage of reduction derived from LOESS smoother and Theil slope. $T = 20$. $\delta_t \sim N(0,\sigma_R^2)$, $\varepsilon_t$ outlier-contaminated. True percentage of reduction: 50 %.

| **Mean** **s.d.** | $\sigma = 0.1$ | | $\sigma = 0.3$ | |
|---|---|---|---|---|
| | LOESS | THEIL | LOESS | THEIL |
| $\sigma_R = 0$ | 49.69 | 49.96 | 47.47 | 49.84 |
| | 5.266 | 1.445 | 19.7 | 4.388 |
| $\sigma_R = 0.03$ | 49.2 | 49.88 | 47.27 | 49.73 |
| | 5.738 | 4.733 | 18.67 | 6.248 |
| $\sigma_R = 0.1$ | 49.72 | 47.92 | 47.06 | 49 |
| | 5.676 | 9.129 | 20.64 | 9.326 |
| $\sigma_R = 0.2$ | 49.33 | 36.6 | 47.49 | 40.72 |
| | 6.444 | 16.37 | 18.33 | 15.23 |

In conclusion, the performance of the methods under consideration is highly dependent on the setting of variance components. For short time series ($T = 10$) and large noise standard deviation ($\sigma = 0.3$), the Theil slope outperforms the LOESS smoother clearly, and only at $\sigma = 0.1$ does the LOESS smoother appear partly to be better

## 3.5 Discussion

Neither the LOESS nor the Theil slope perform satisfactorily under all settings. Both methods have specific disadvantages. The Theil slope does not work properly with highly non-linear trends and breaks down if the decline stops after about 30 % of the period of time. The LOESS smoother is highly efficient as long as the Normal distribution holds, but it is sensitive to outlier contamination. If the risk of outliers or temporal heteroscedasticity (e.g., due to climate variability) cannot be avoided, estimates derived from the LOESS smoother in its current specification should not be used without outlier inspection. Only for short time series does the estimator derived from the Theil slope seem to be an acceptable alternative. For longer time series, neither the Theil slope nor the LOESS smoother can be generally recommended for estimating reduction rates.

## 3.6 Acknowledgement

This material was prepared by Steffen Uhlig, Germany.

## 3.7 References

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

## ANNEX 2

## MAPPING OF HARMFUL EVENTS RELATED TO PHYTOPLANKTON BLOOMS IN ICES MEMBER COUNTRIES

This mapping exercise provides a broad geographical overview of harmful events related to phytoplankton blooms in the ICES area for the ten-year period 1989–1998. The work was carried out through the ICES/IOC Working Group on Harmful Algal Bloom Dynamics (WGHABD), under P. Gentien (IFREMER, Brest, France) as Chair. Data, contributed by WGHABD participants, were processed by C. Belin (IFREMER, Nantes, France) and the final maps were generated by B. Raffin (IFREMER, Nantes, France) using ArcInfo© software.

The maps indicate the presence of toxins or observations of animal/plant mortality if detected, regardless of the level of toxicity. The maps also show regular monitoring sites and give an indication of the frequency of harmful bloom events during the ten-year period. Seven different types of events were considered: amnesic shellfish poisoning (ASP); ciguatera fish poisoning (CFP); diarrhetic shellfish poisoning (DSP); neurotoxic shellfish poisoning (NSP); paralytic shellfish poisoning (PSP); other toxic effects, such as cyanobacterial toxin poisoning; and animal/plant mortality. Each type of event is presented in a separate figure.

The maps can also be seen on the ICES website at http://www.ices.dk/status/decadal/dec8998.htm.

**GENERAL DISCLAIMER–WARNING**

These maps present information on toxin presence from 1989–1998. The information is based on annual national reports by ICES Member Countries. The information available on individual events varies greatly from event to event and from country to country. The monitoring intensity, number of monitoring stations, number of samplings, etc., also vary greatly and, therefore, there is not a direct proportionality between recorded events and actual occurrences of, e.g., toxicity in a given region. Furthermore, areas with numerous recorded occurrences of harmful algal events (HAEs), but with efficient monitoring and management programmes, may have very few problems and a low risk of intoxication, whereas rare HAEs in other areas may cause severe problems and represent significant health risks.

Therefore, these maps should be interpreted with caution with regard to the risk of intoxication by seafood products from the respective areas/regions/countries.

IOC and ICES are not liable for the possible misuse of this information.

**Figure A2.1.** ICES Member Countries in Europe are indicated by gray shading. Regular monitoring of bloom events in 1998 is carried out in the areas marked by a heavy black line.

**DISCLAIMER – WARNING**
HAEDAT maps should be interpreted with caution regarding risk of intoxication by seafood products from the respective areas/regions/countries. The IOC and ICES are not liable for possible misuse of this information.

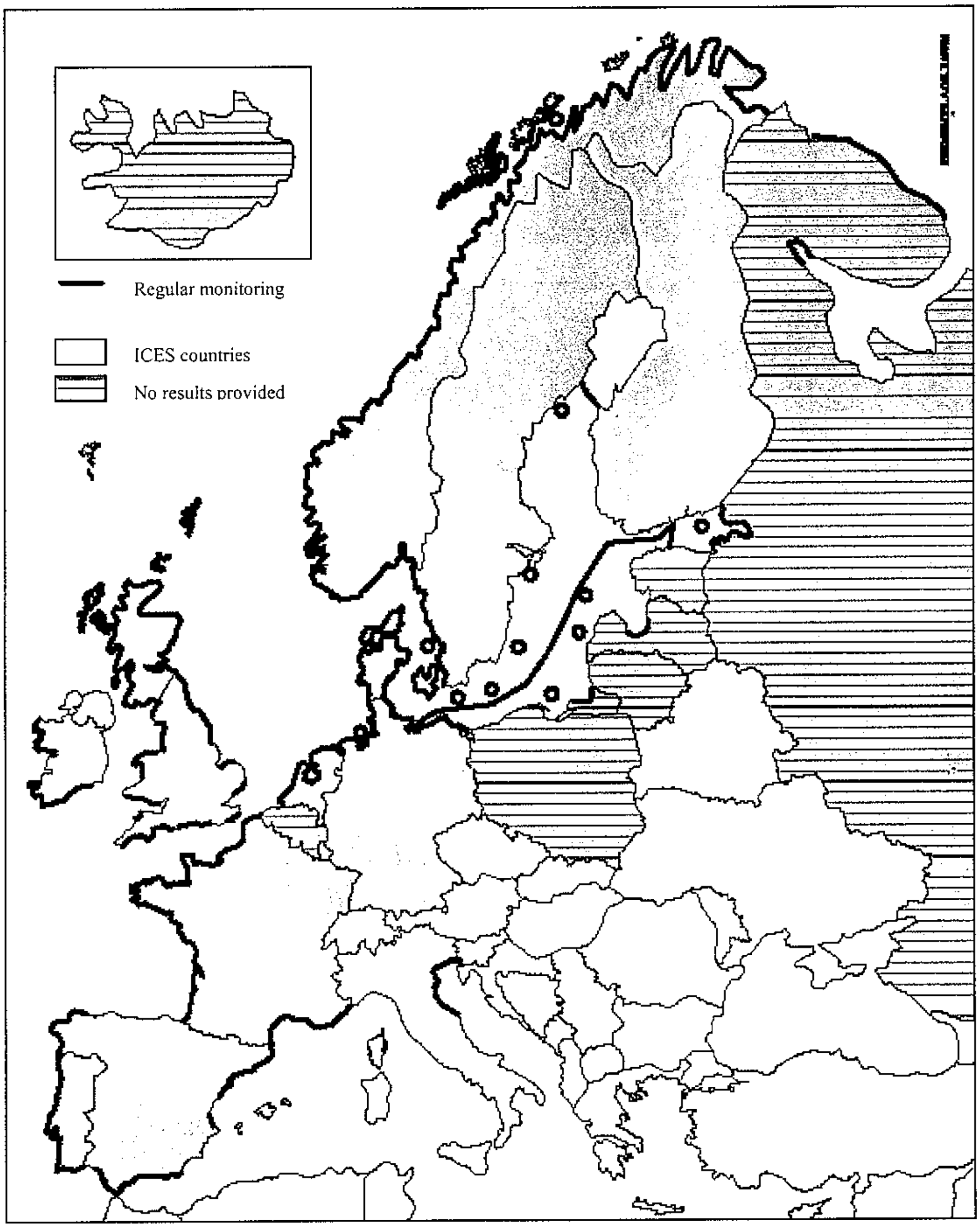

**Figure A2.2.** The occurrence and frequency of diarrhetic shellfish poisoning (DSP) events in Europe (1987–1998) are indicated by the black circles.

**DISCLAIMER – WARNING**
HAEDAT maps should be interpreted with caution regarding risk of intoxication by seafood products from the respective areas/regions/countries. The IOC and ICES are not liable for possible misuse of this information.

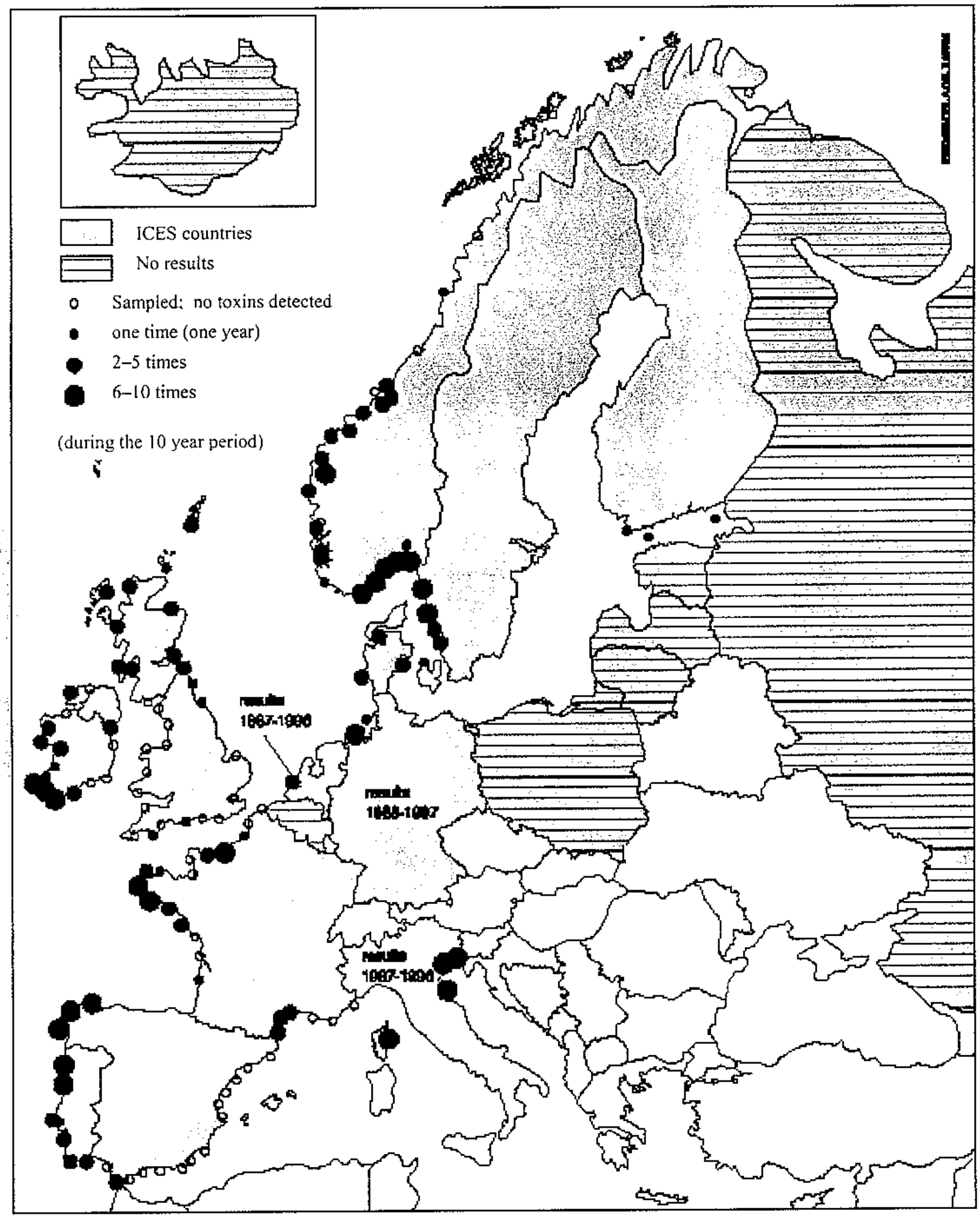

**Figure A2.3.** The occurrence and frequency of paralytic shellfish poisoning (PSP) events in Europe (1987–1998) are indicated by the black circles.

**DISCLAIMER – WARNING**
HAEDAT maps should be interpreted with caution regarding risk of intoxication by seafood products from the respective areas/regions/countries. The IOC and ICES are not liable for possible misuse of this information.

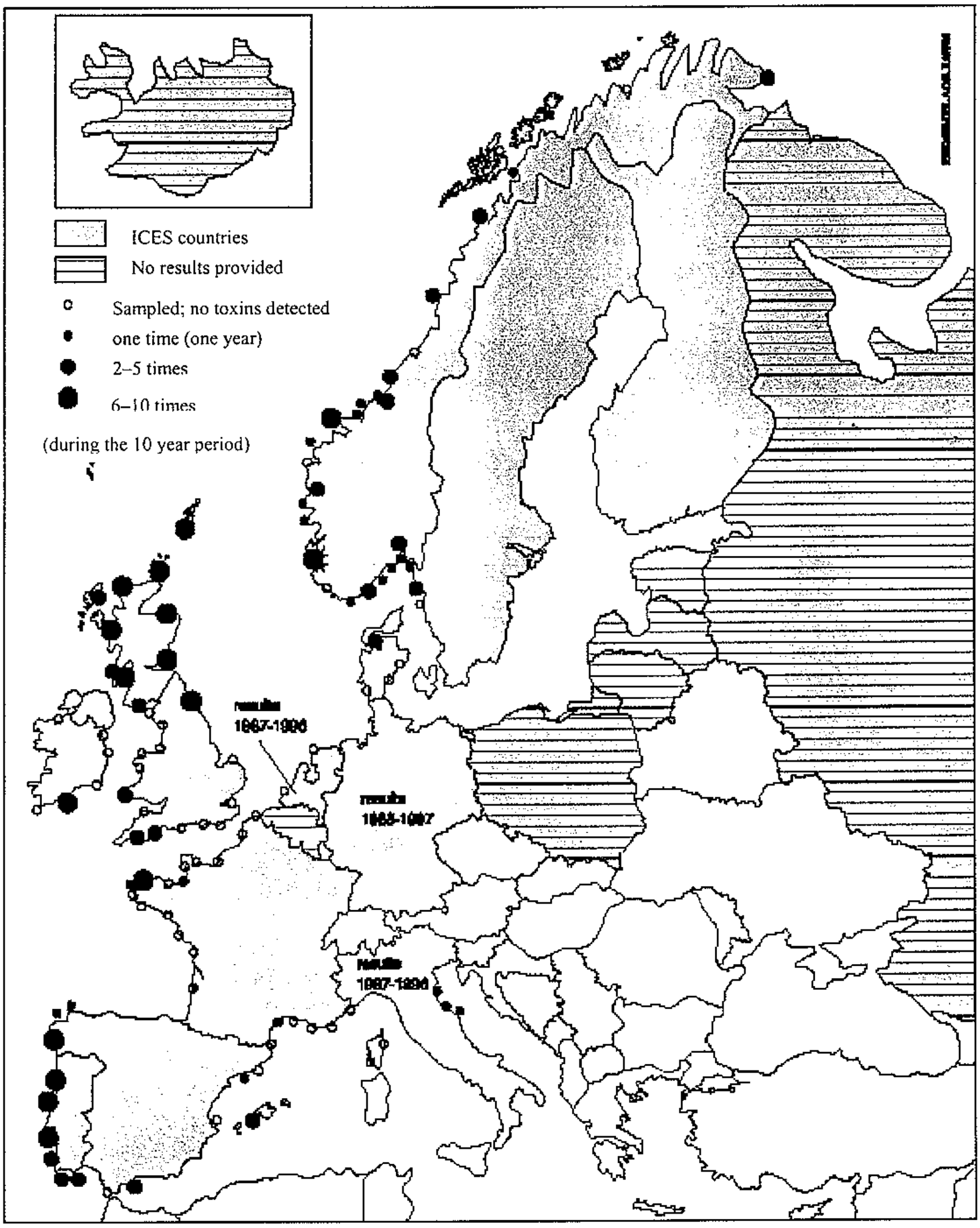

**Figure A2.4.** The occurrence and frequency of amnesic shellfish poisoning (ASP) events in Europe (1987–1998) are indicated by the black circles.

**DISCLAIMER – WARNING**
HAEDAT maps should be interpreted with caution regarding risk of intoxication by seafood products from the respective areas/regions/countries. The IOC and ICES are not liable for possible misuse of this information.

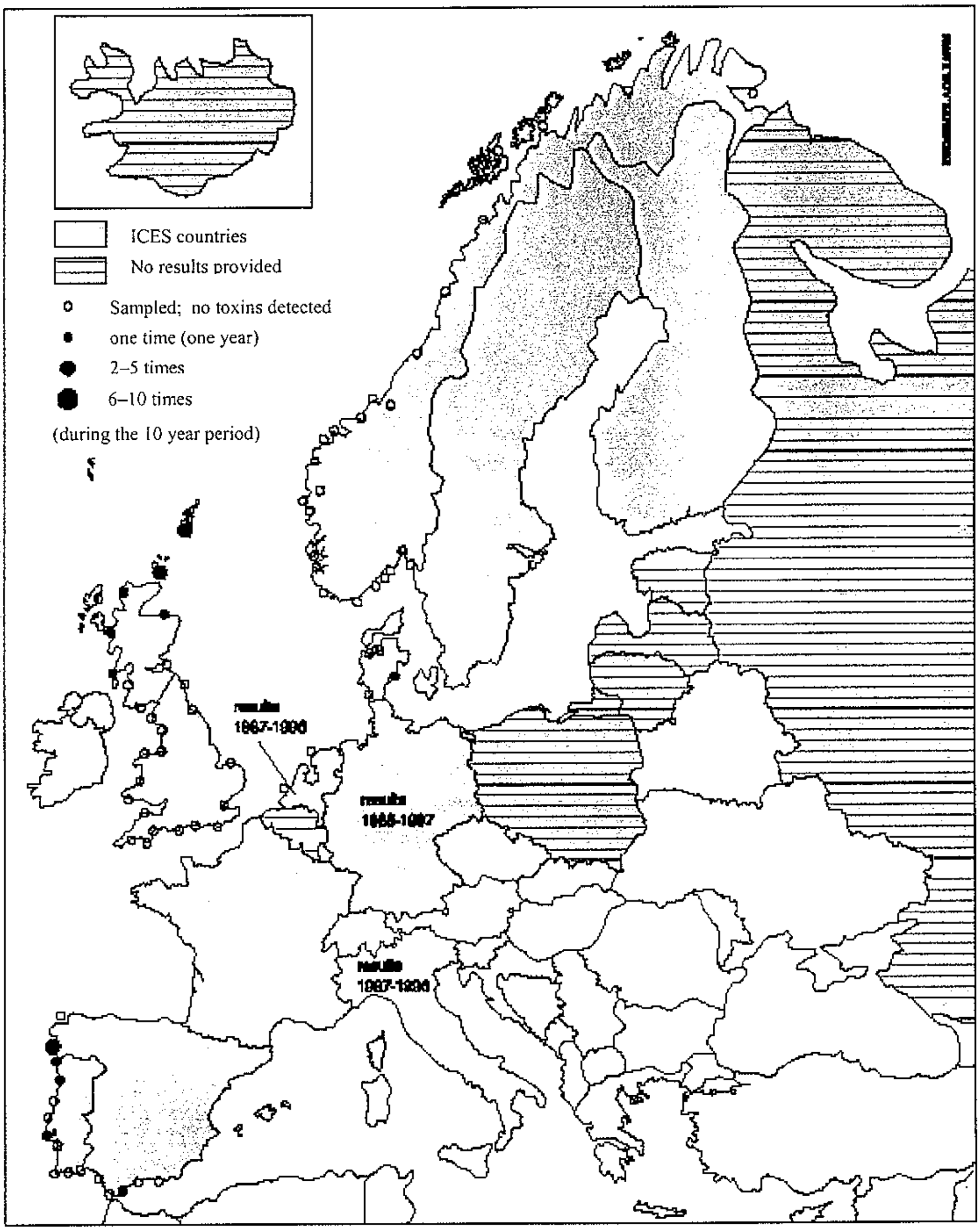

**Figure A2.5.** The occurrence and frequency of animal or plant mortalities in Europe (1987–1998) are indicated by the black circles.

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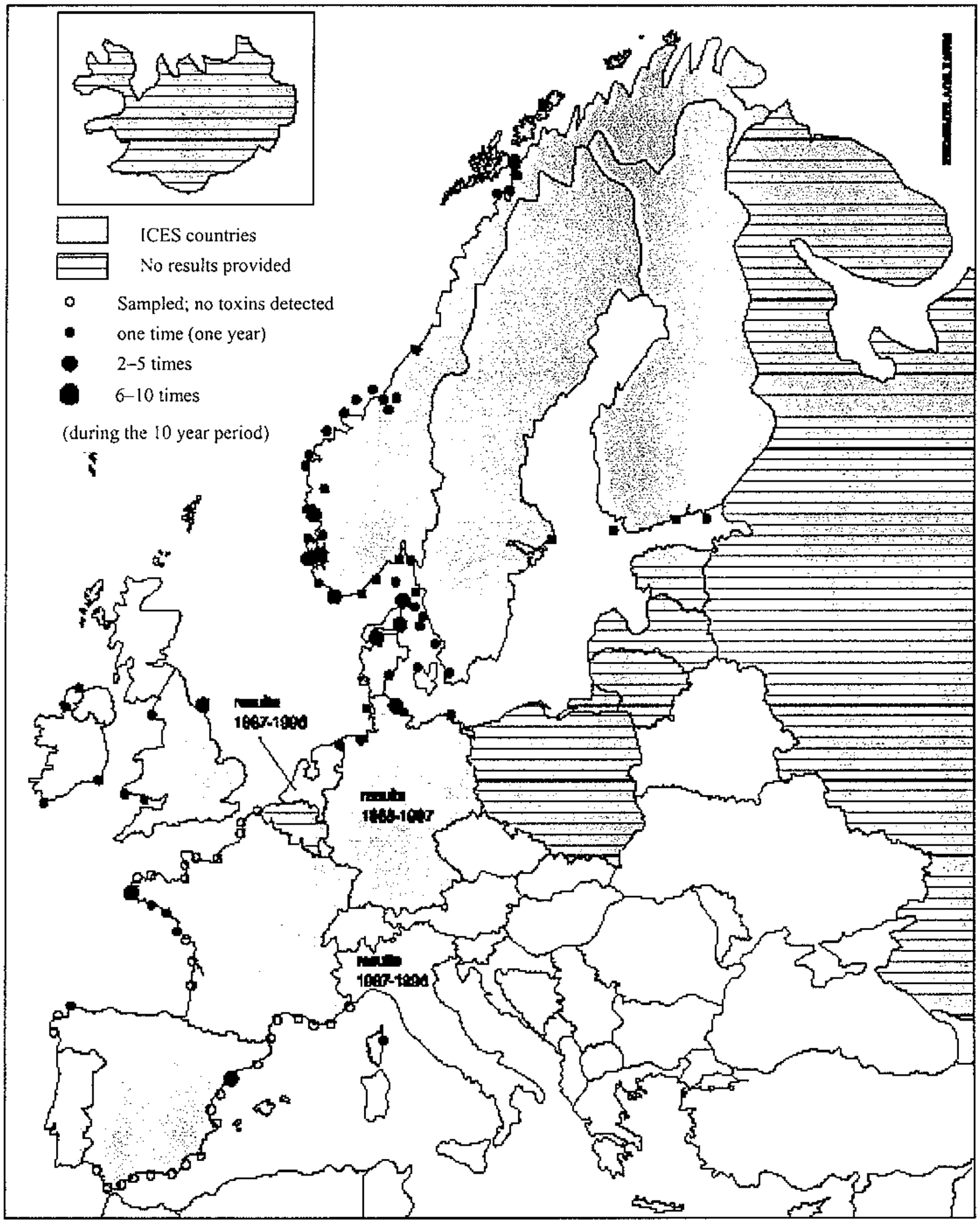

**Figure A2.6.** The occurrence and frequency of other toxic effects, such as cyanobacteria toxicity, in Europe (1987–1998) are indicated by the black circles.

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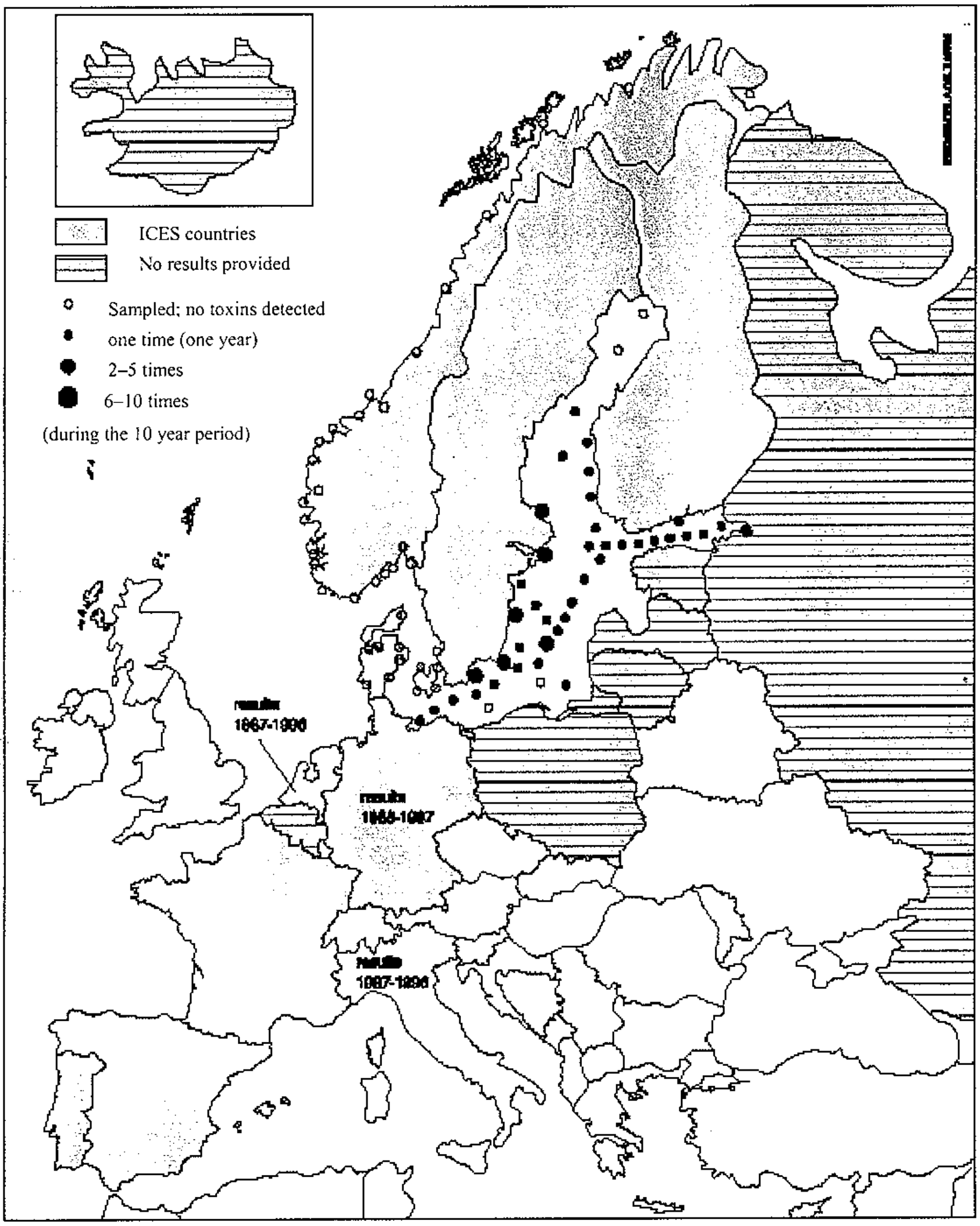

**Figure A2.7.** ICES Member Countries in North America are indicated by gray shading. Regular monitoring of bloom events in 1998 is carried out in the areas marked by a heavy black line.

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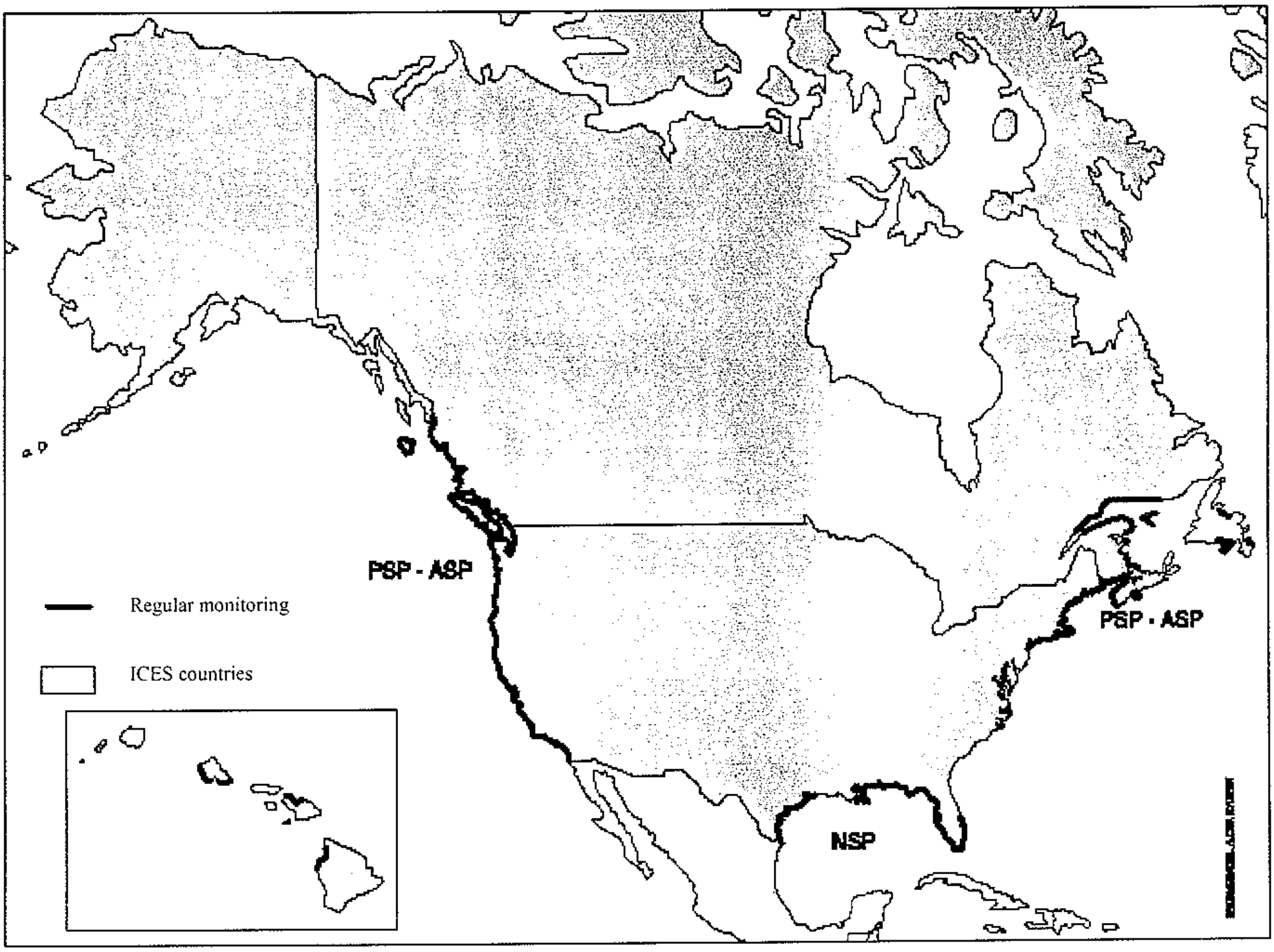

**Figure A2.8.** The occurrence and frequency of paralytic shellfish poisoning (PSP) events in Canada (1989–1998) and in the United States (1988–1997) are indicated by the black circles.

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HAEDAT maps should be interpreted with caution regarding risk of intoxication by seafood products from the respective areas/regions/countries.
The IOC and ICES are not liable for possible misuse of this information.

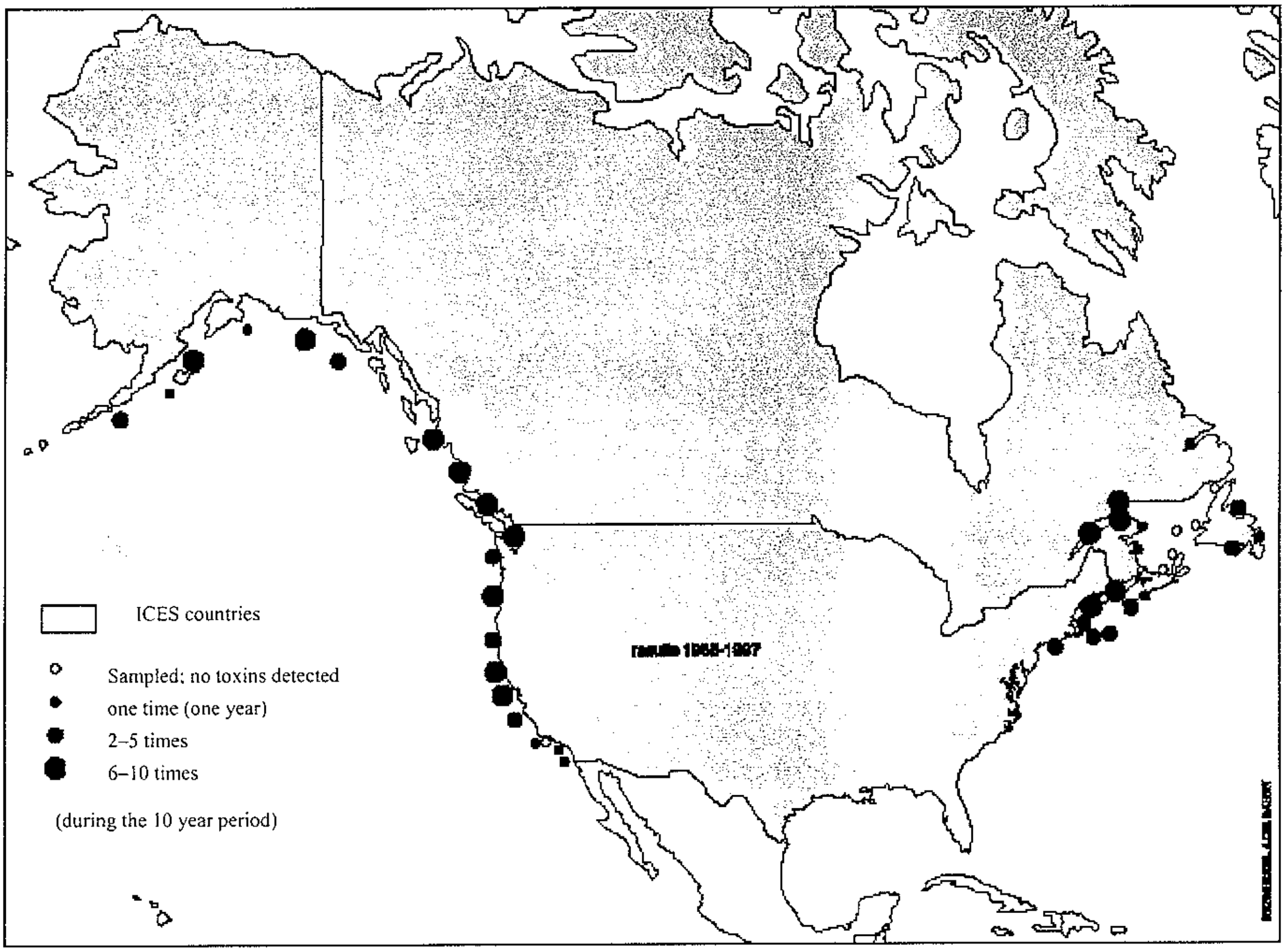

**Figure A2.9.** The occurrence and frequency of neurotoxic shellfish poisoning (NSP) events in the United States (1988–1997) are indicated by the black circles.

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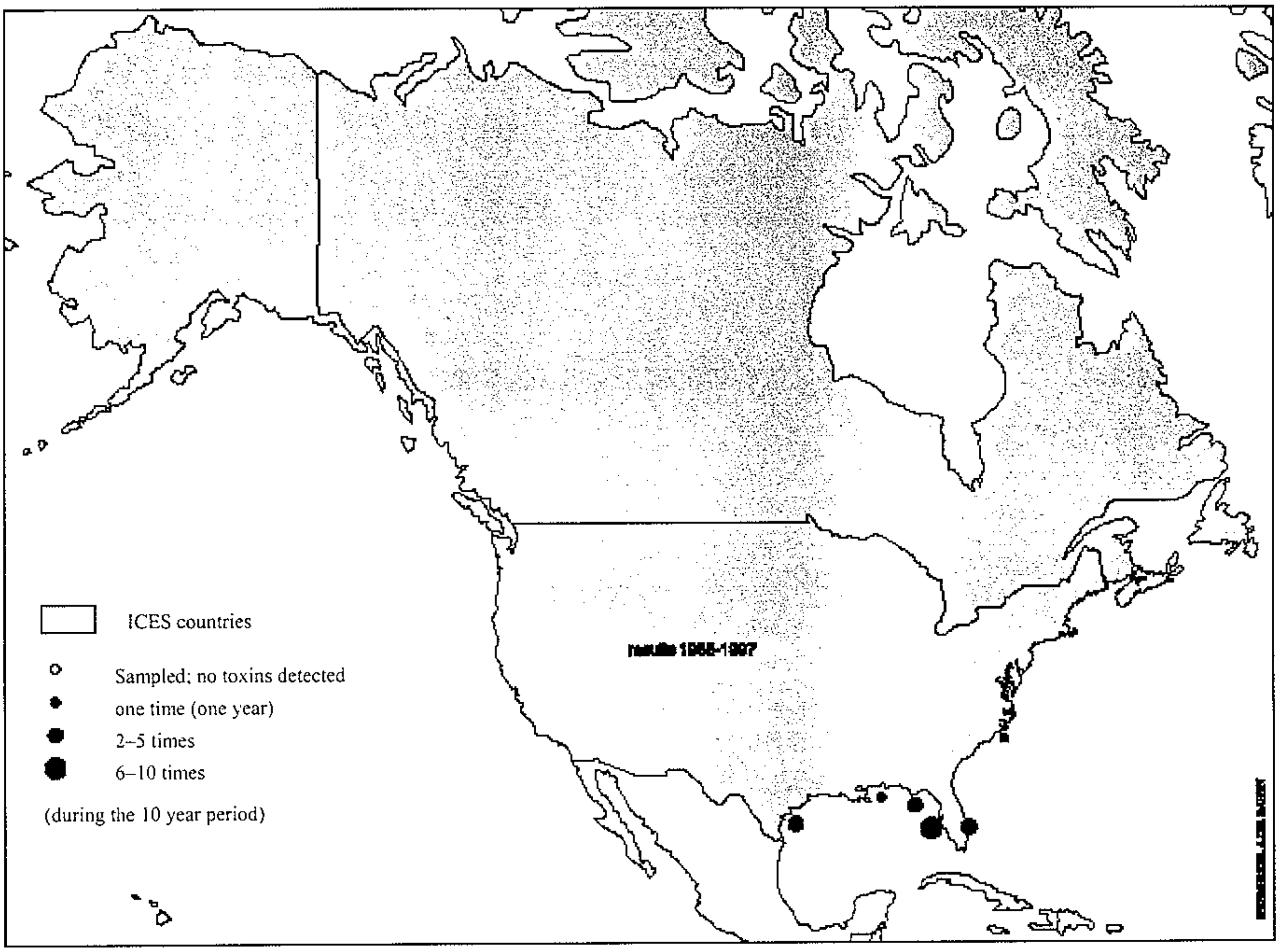

**Figure A2.10.** The occurrence and frequency of amnesic shellfish poisoning (ASP) events in Canada (1989–1998) and in the United States (1988–1997) are indicated by the black circles.

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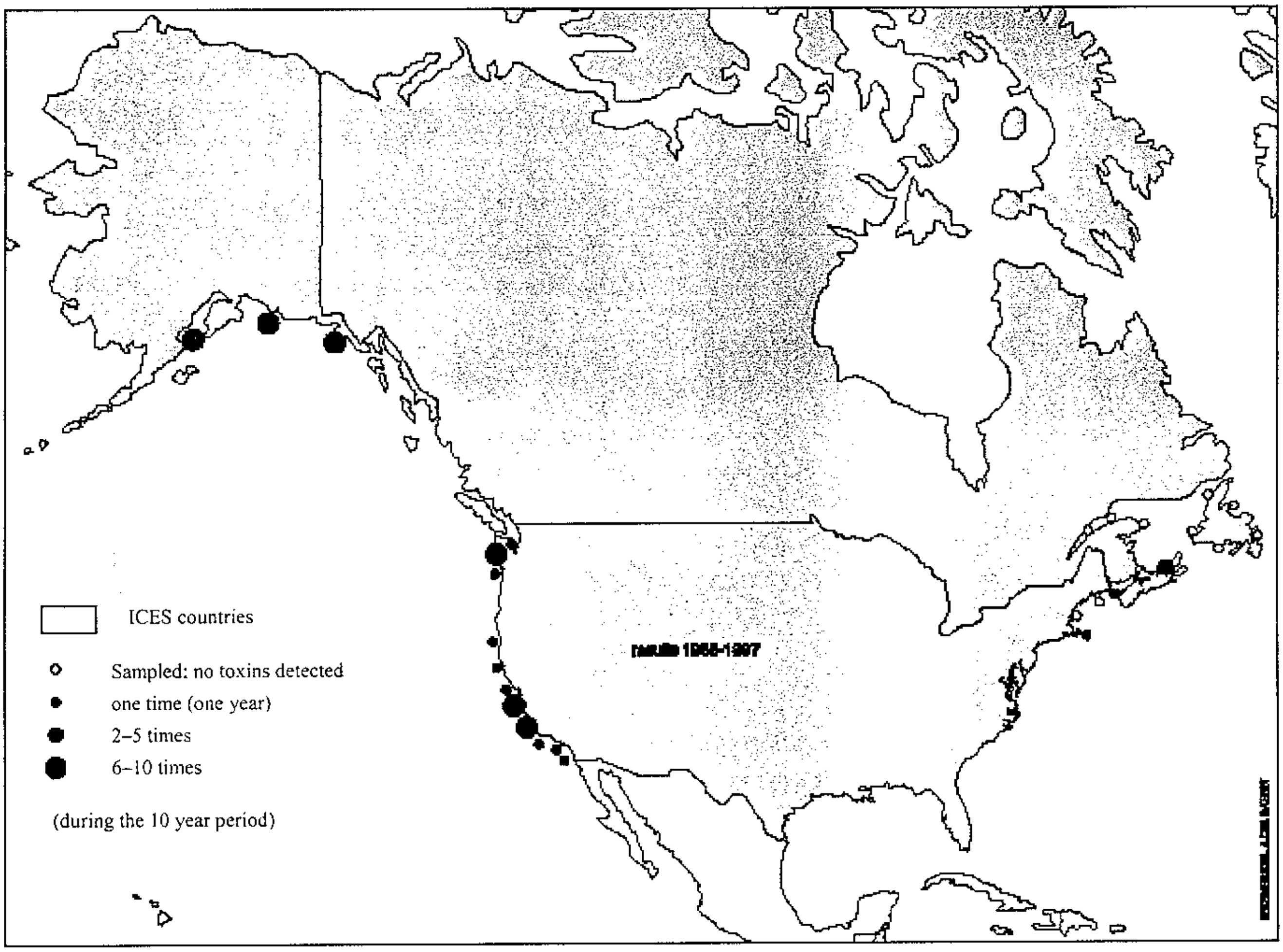

**Figure A2.11.** The occurrence and frequency of animal or plant mortalities in Canada (1989–1998) and in the United States (1988–1997) are indicated by the black circles.

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HAEDAT maps should be interpreted with caution regarding risk of intoxication by seafood products from the respective areas/regions/countries. The IOC and ICES are not liable for possible misuse of this information.

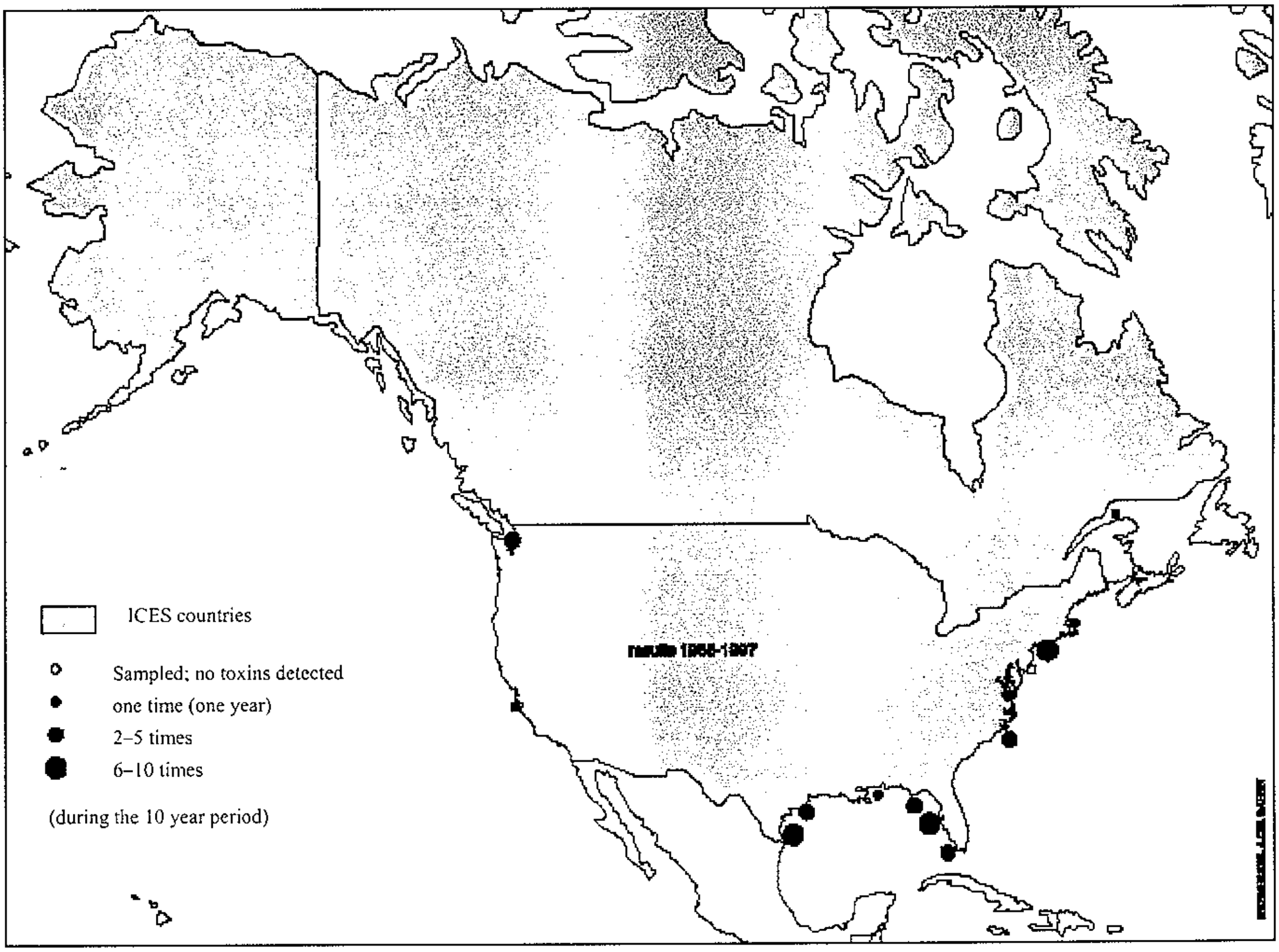
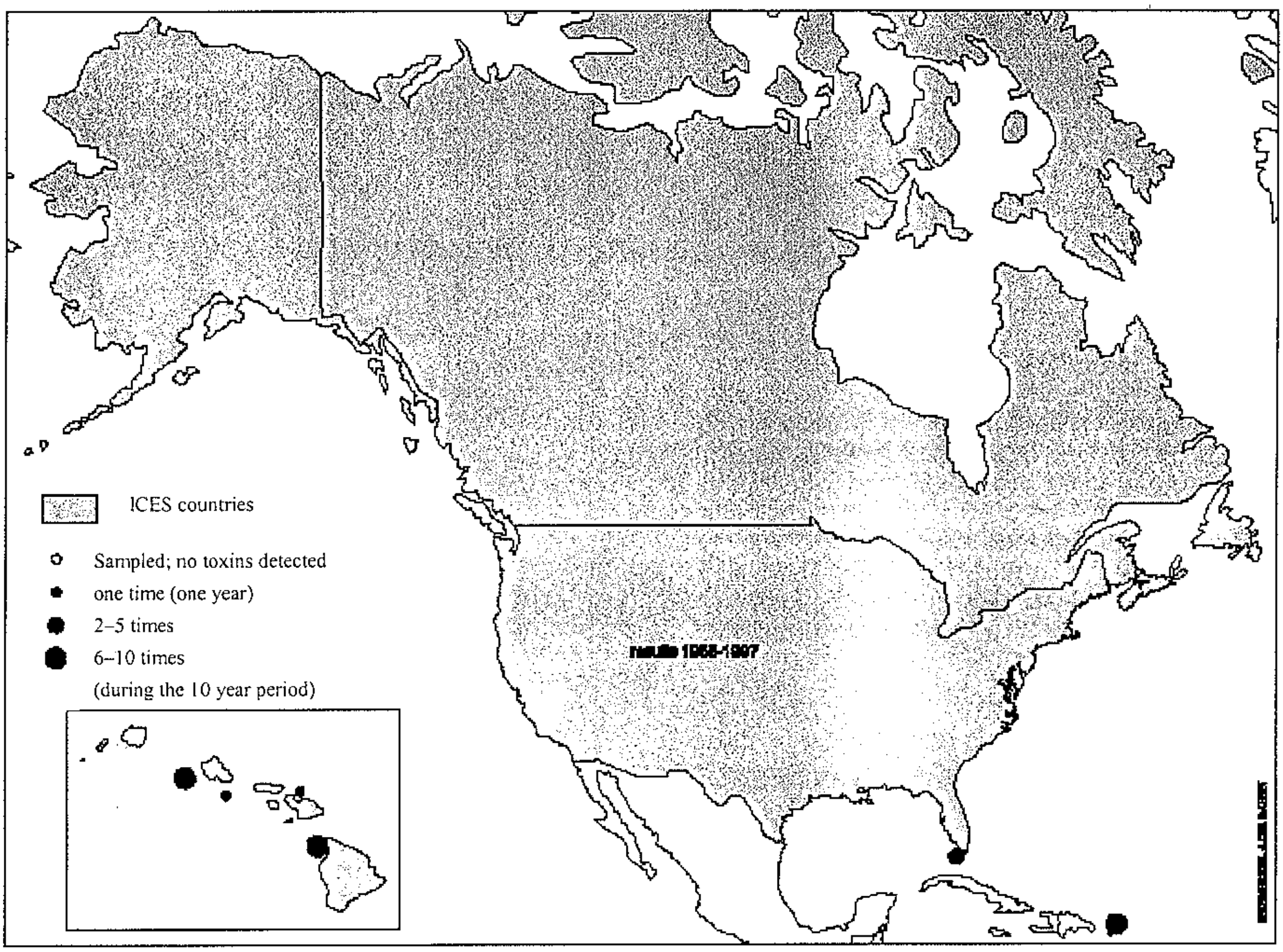


**Figure A2.12.** The occurrence and frequency of ciguatera fish poisoning (CFP) in the United States (1988–1997) are indicated by the black circles.

**DISCLAIMER – WARNING**
HAEDAT maps should be interpreted with caution regarding risk of intoxication by seafood products from the respective areas/regions/countries. The IOC and ICES are not liable for possible misuse of this information.



## ANNEX 3

## DISTRIBUTION AND EFFECTS OF TRIBUTYLTIN—AN UPDATE TO MID-1999


### ABSTRACT

The focus of this update to earlier reviews of the environmental fate and effects of tributyltin (TBT) derived from antifouling preparations is timely, as it comes ten to fifteen years after controls were instituted on the use of TBT on small vessels less than 25 m in length. During this period, numerous studies have documented the environmental improvements in estuaries and enclosed waters which followed the return to copper-based antifouling preparations, albeit now often incorporating 'booster biocides' such as Irgarol 1051 and Diuron (Law, 1997; Thomas, 1998). Attention has also focused increasingly on the impact of the continued use of TBT paints on larger sea-going vessels, and it is in the light of this continued interest that the present review note has been prepared. The intention is to summarize the current worldwide knowledge of the fate and distribution (and, where possible, effects) of TBT in open sea areas, shipping lanes, and close to ports and harbours frequented by large vessels.


### 1 DISTRIBUTION OF BUTYLTINS IN SEA WATER AND MUSSELS

Studies of the distribution of butyltins along a heavy tanker route in the Strait of Malacca in 1996 indicated recent inputs, with a tributyltin (TBT) to dibutyltin (DBT) ratio of greater than two (Hashimoto *et al.*, 1998) (see Table A3.1). In open ocean areas away from the shipping route, butyltins were generally undetectable. The distribution pattern was consistent with that observed for oil slicks resulting from oil tanker traffic along that route, suggesting that such large vessels represented the major source. Studies undertaken in 1996 by Swennen *et al.* (1997) of several species of sublittoral gastropods also found that imposex prevalence in those animals was related to their proximity to shipping lanes in the Strait of Malacca and the Gulf of Thailand. Hashimoto *et al.* (1998) also reported studies conducted in Tokyo Bay in 1993, 1994, and 1996. Higher concentrations of TBT were observed in this inshore area (up to approximately 11 ng $l^{-1}$), although concentrations of monobutyltin (MBT) and DBT were substantially higher than concentrations of TBT, probably as the result of wastewater discharges containing those compounds.

The decrease in concentrations of butyltin compounds in French waters observed following the 1982 ban has now ceased, with current concentrations remaining above the toxicity threshold of 1 ng $l^{-1}$ (Michel and Averty, 1999). The overall concentration range observed for TBT was 1.7 to 280 ng $l^{-1}$, with high concentrations being recorded in some ports and marinas. For marinas, the highest concentrations were observed in the eastern Channel and eastern Mediterranean coasts, up to 88 ng $l^{-1}$ and 199 ng $l^{-1}$, respectively. Lower concentrations were observed in marinas located along the Atlantic, western Channel and western Mediterranean coasts, rarely exceeding 20 ng $l^{-1}$. Port areas continue to be major sources of contamination, with continuing harmful effects on both natural (Huet *et al.*, 1996) and cultivated mollusc populations.

Mussels collected at a control site in the St. Lawrence Estuary in Canada (at the Bic Provincial Park) contained low concentrations of organotins, but they were detected at all sites. This suggests low-level TBT contamination of the seawater body, which circulates along the south shore of the St. Lawrence Estuary during the summer (Saint-Louis *et al.*, 1997). Butyltins were determined in sediments and mussels (*Mytilus edulis*) collected in 1995/1996 from sites in the Gulf of St. Lawrence (Saint-Jean *et al.*, 1999). TBT concentrations in mussels ranged from below the limit of detection to 671 µg $kg^{-1}$ Sn dry weight, and the highest DBT concentration observed was 378 µg $kg^{-1}$ Sn dry weight. At most sampling sites there was no clear relationship between butyltin concentrations in sediments and mussels. The authors concluded that TBT inputs were still well above expected levels after eight years of regulation on the use of TBT-based paints. The major source of TBT was ascribed to pleasure boating activities, with a lesser but chronic contribution from commercial vessels.

**Table A3.1.** Concentrations of butyltin species in waters.

| Location | Year | TBT (ng $l^{-1}$) | DBT (ng $l^{-1}$) | MBT (ng $l^{-1}$) | Reference |
|---|---|---|---|---|---|
| France | 1997 | 1.7 to 280 | – | – | Michel and Averty (1998) |
| Strait of Malacca | 1996 | 0.1 to 5.2 | < 0.1 to 2.4 | 0.2 to 5.9 | Hashimoto *et al.* (1998) |
| Baltic Sea | 1994/1995 | < 10 to 210 | < 10 to 72 | < 10 to 65 | Kalbfus *et al.* (1996) |
| North Sea | 1994/1995 | < 10 to 320 | < 10 to 120 | < 10 to 110 | Kalbfus *et al.* (1996) |

## 2 IMPOSEX IN DOGWHELK POPULATIONS

The incidence of imposex in dogwhelk populations has been used extensively as a biomonitor of TBT levels in sea water. This phenomenon is fully developed at ambient TBT concentrations of 1–2 ng $l^{-1}$, and females are fully sterilized at concentrations above 5 ng $l^{-1}$ (Gibbs and Bryan, 1996; Matthiessen and Gibbs, 1998). In the section below, all the dogwhelks studied are *Nucella lapillus* unless otherwise indicated.

In a recent study of imposex along the coasts of southern and southwestern Wales, samples were collected in October 1996 and August 1997 between Milford Haven and Barry in the Severn estuary. The study found that the highest levels of TBT were confined to shipping areas (Morgan *et al.*, 1998). While none of these populations were sterile, many of the samples close to ports exhibited levels of imposex indicating TBT levels well above the UK Environmental Quality Standard (EQS) of 2 ng $l^{-1}$. In addition, the authors noted that ports in the Shetlands, Norway, the Tyne Estuary, and Cork Harbour, which are used by large commercial vessels, are known hot-spots of TBT pollution as indicated by the incidence of imposex in dogwhelks (Minchin *et al.*, 1996; Moore *et al.*, 1996; Morgan *et al.*, 1998). Sullom Voe in the Shetlands is a large oil terminal that is served by a number of offshore pipelines, and is also visited by a large number of oil tankers (700–900 per year) using TBT-based antifoulants (Atkins, 1998). The Voe is a narrow seaway with a restricted water flow and water exchange. There are no dockyard facilities for other vessels, and there is no significant pleasure boat or fish farming activity in the area. Therefore, any TBT present in the water of the Voe is derived from visiting tankers. In 1987, female dogwhelks in the Voe suffered from severe imposex, with up to 90 % of females around the loading jetties having blocked reproductive tracts (Davies and Bailey, 1991). The reproductive capacity of the populations in the Voe decreased between 1987 and 1990; in 1987 47 % of females sampled were sterile, while in 1990 this had risen to 65 %. All females sampled at the terminal site were sterile, and it seems that dogwhelk populations within the Voe are now maintained by recruitment from adjacent areas (Atkins, 1998).

In 1992, studies were conducted at 45 locations in Icelandic waters and imposex was found to occur in dogwhelks at 38 of these sites. The level of imposex was highest in the vicinity of larger harbours (such as Reykjavik and Hafnarfjordur) and the effects could be observed almost 18 km away (Svavarsson and Skarphedinsdottir, 1995). Similar effects have been observed in 41 populations of dogwhelks along the Norwegian coast (Følsvik *et al.*, 1998). Dogwhelks near large port facilities (such as Stavanger, Florø, and Måløy) were more heavily affected than those in other areas. The concentrations of TBT in dogwhelk tissue were found to correlate with the degree of imposex.

Minchin and Minchin (1997) assessed imposex in dogwhelks from 66 locations in Bantry Bay, southwestern Ireland in 1996. The highest imposex levels were found in the harbour of the fishing port of Castletownbearhaven, which is the second largest fishing port in southwestern Ireland. Stations within 600 m of the centre of the harbour were devoid of dogwhelks, although they had been common at some of these sites in the 1960s. Dogwhelks beyond this zone showed a high incidence of imposex, which declined with distance from the source. The major input in this case was from fishing vessels of 25 m to 50 m in length, which can still be legally painted with TBT-based antifoulants.

Prouse and Ellis (1997) studied the incidence of imposex in dogwhelks from harbours in eastern Canada, and found Halifax Harbour, Nova Scotia, to be the most severely affected. Six of the eleven sites surveyed yielded dogwhelks with imposex frequencies between 65 % and 100 %, and dogwhelks were absent from the other five sites, although they appeared to provide a good habitat. These five sites were located within the inner harbour close to marinas, shipyards, and docking facilities for large vessels, and this grouping was considered consistent with their forming a central block too contaminated with TBT and possibly other chemicals for *Nucella* to survive. The lack of imposex and TBT in dogwhelks at Argentia, a decommissioned United States naval base in Newfoundland, suggested that the current source of TBT in Halifax Harbour and the other ports studied was the continuing release from large vessels using TBT-based antifoulants, which is not currently controlled.

The incidence of imposex in populations of the dogwhelk *Lepsiella scobina* was compared before and after restrictions were introduced on the use of TBT-based antifouling paints in New Zealand, and two areas surveyed in 1988/1989 were restudied in 1994/1995. In an area subject only to inputs from pleasure craft (Porirua Inlet), there was a significant decline in the incidence of imposex over this period; but this was not mirrored in Wellington Harbour, which handles both commercial and pleasure traffic (Smith, 1996).

As a result of these studies, it is evident that where large numbers of commercial vessels use a restricted waterway, it is very likely that local dogwhelk populations will exhibit significant levels of imposex. This indicates that in the vicinity of many major ports and harbours in Europe (and, presumably, other parts of the world) TBT inputs from large shipping will be significant, irrespective of other sources of TBT (Atkins, 1998). Dogwhelks near these installations are therefore likely to be affected by TBT for the foreseeable future, unless additional controls on its use for commercial vessels are implemented.

## 3 IMPOSEX IN WHELK POPULATIONS

Two species of whelks (*Buccinum undatum* and *Neptunea antiqua*) collected from three sites in western Scotland in 1997 showed varying degrees of imposex (Połoczanska and Ansell, 1999). Both the incidence of imposex and the degree of imposex development were greater in *N. antiqua* than in *B. undatum*. The results probably reflect long-term contamination of sediments by TBT originating from inputs prior to the 1987 ban. The levels of imposex observed were not such as to affect the fishery potential of these species in western coastal waters of Scotland.

A recent study in the southern North Sea has demonstrated imposex in whelks *Buccinum undatum* resulting from TBT contamination from shipping (Ten Hallers-Tjabbes *et al.*, 1994), with a peak of 100 % incidence in whelks sampled close to the entrance channel to Rotterdam Harbour. It is also considered likely that the use of TBT in antifouling paints resulted in the local extinction of whelks within the Dutch Wadden Sea following their depletion by fishing (Cadée *et al.*, 1995; Mensink *et al.*, submitted). These authors suggested that a failure to control the continued use of TBT and the absence of fishery-free zones in the North Sea would cause the extinction of whelks there also, although their interpretation has been criticized by others on the grounds that the observed effects cannot be linked unequivocally to TBT (ORTEPA, 1996).

The decline of North Sea whelks between the late 1960s to early 1970s and the early 1990s has been documented by Ten Hallers-Tjabbes *et al.* (1996). In the Eastern Scheldt, nearly all female common whelks (more than 91 %) showed a degree of imposex (more than half of them the advanced stages), although the whelks did reproduce in 1994 and 1995 (Mensink *et al.*, 1997b).

Despite the ban on the use of TBT in antifouling paints for vessels below 25 m in length in 1990, the incidence of imposex has not been reduced in this area in subsequent years. Some continued use of TBT in marinas has been indicated by monitoring of concentrations in water (Ritsema, 1994), and this area is subject to agricultural inputs of triphenyltin as well as TBT release from large vessels. However, whole body levels of triphenyltin in whelks from the Eastern Scheldt did not show a significant difference between females with advanced stages of imposex and visually unaffected females (Mensink *et al.*, 1997a).

It should be noted that, whereas dogwhelks live intertidally and mainly on hard substrates and, thus, are affected predominantly by coastal inputs of TBT, the whelk *Buccinum undatum* is found subtidally to 200 m depth (Ten Hallers-Tjabbes and Boon, 1995). Thus, in the open sea these animals will be affected primarily by inputs from passing shipping traffic.

## 4 TBT IN MARINE MAMMALS

Recent findings of TBT in the tissues (particularly liver) of marine mammals in nearshore areas and the open ocean is a cause for concern (Tanabe *et al.*, 1998; Kannan and Falandysz, 1997, 1999; Kannan *et al.*, 1998; Law *et al.*, 1998). The accumulation of TBT in the tissue of marine mammals has previously been demonstrated for animals from the Indian and Pacific Oceans and, particularly, for finless porpoise (*Neophocoena phocanoides*) from the Seto-inland Sea in Japan (Iwata *et al.*, 1995). Recent studies of animals stranded in the UK have shown detectable concentrations of butyltins in both coastal and pelagic marine mammals, the latter animals feeding over the continental shelf edge and slope, and in the deep oceanic waters of the Atlantic Ocean (Law *et al.*, 1998; Law *et al.*, 1999). The highest concentration of total butyltin ($\Sigma BT = MBT + DBT + TBT$) found in harbour porpoise (*Phocoena phocoena*) liver was 640 µg $kg^{-1}$ wet weight, while in grey seals (*Halichoerus grypus*) ΣBT concentrations were much lower, indicating either that the seals have a lower dietary intake of butyltins or a higher catabolic capacity for these compounds than porpoises. A similar order of concentration (up to 312 µg $kg^{-1}$ wet weight) was observed for sixteen pelagic marine mammals (two mysticete and ten odontocete species) analysed. These included a fin whale, a minke whale, and two species of beaked whales. In both studies, about 20 % of the total butyltin burden in liver was TBT, with DBT as the major component. Butyltins were also found in tissues of sperm whales (*Physeter macrocephalus*) stranded on the Dutch and Danish coasts (Ariese *et al.*, 1998), although these animals also live and feed exclusively in the deep ocean.

Southern sea otters (*Enhydra lutris nereis*) found dead along the coast of California contained total butyltins in the range 40–9200 µg $kg^{-1}$ wet weight, which varied depending on location and gender (Kannan *et al.*, 1998). At locations close to harbours handling large vessels legally painted with organotin antifoulants, TBT predominated over its degradation products DBT and MBT, suggesting recent exposure. Otters collected within enclosed marinas exhibited concentrations at least ten times higher than those from open locations. Mean concentrations of ΣBT in female sea otter livers (1420 µg $kg^{-1}$ wet weight) were almost two-fold greater than those in males (750 µg $kg^{-1}$ wet weight), possibly due to their higher feeding rates and their smaller body size. Otters affected by infectious diseases also showed higher concentrations of ΣBT (mean = 1570 µg $kg^{-1}$ wet weight) than those which died of trauma or unknown causes (mean = 220 µg $kg^{-1}$ wet weight), which is consistent with an immunosuppressive effect.

These data demonstrate the widespread distribution of butyltin residues in deep offshore waters and oceanic food chains, but further studies are needed in order to elucidate the potential impact of butyltin body burdens on individual marine mammals and populations if TBT inputs from large vessel traffic continue as at present.

## 5 DISPOSAL OF TBT-CONTAMINATED DREDGINGS

Concentrations of TBT and its degradation products DBT and MBT are often very high in sediments (see Table A3.2), particularly those in areas frequented by shipping or where inputs from pleasure boats were high, as degradation of TBT is slow. The accumulation of TBT and its degradation products (MBT and DBT) in sediments can result from two processes. Firstly, TBT can adsorb onto sediment particles from the dissolved phase. These organotin species can then be subject to degradation and desorption processes. These processes are strongly influenced by the physicochemical characteristics of the sediment, such as grain size and organic matter content. Secondly, close to harbours, dry docks and associated activities, paint flakes containing TBT can be deposited directly to sediments following release during the removal of antifouling paints from vessels prior to repainting. This often involves sandblasting or high-pressure water hoses, and the recovery of wastes generated in this way should be mandatory (Michel and Averty, 1999).

Stronkhorst (1996) studied the effects of TBT use on ships and in dockyards around Rotterdam, and measured the concentrations of TBT in coastal waters. He emphasized the importance of sediments as both a sink and a source for TBT, since a major source of coastal contamination resulted from the disposal of dredged harbour sediments at disposal sites some distance offshore. Stronkhorst *et al.* (1995) determined an ecological criterion of 500 μg $kg^{-1}$ dry weight for TBT in sediments, above which environmental harm would result. This value was exceeded in some samples from both Amsterdam and Rotterdam harbours in 1994/1995, the concentration ranges observed being 30 to 2000 μg $kg^{-1}$ dry weight and 10 to 2100 μg $kg^{-1}$ dry weight, respectively, for the two locations (Stronkhorst, 1996). Stewart and Thompson (1997) determined butyltin concentrations in a series of five sediment cores from harbours and marinas in Canada. A sediment core taken within a marina showed a maximum TBT concentration at 8 cm depth, indicating a reduction in TBT flux to the surface sediments following a ban on the use of TBT paints on vessels of less than 25 m in length. However, no such trend was visible in harbour sediment cores, with the highest concentrations of TBT in superficial sediments. The presence of TBT throughout the sediment core implies that it must be considered a persistent contaminant, and the profiles observed in harbour sediment cores do not suggest any amelioration of TBT burdens at these sites following the partial ban in TBT usage. In Halifax Harbour, TBT concentrations in sediments from the inner harbour have also increased between 1988 and 1994 (Prouse and Ellis, 1997).

In southwestern Spain, TBT was found to be the predominant species in water, with the highest concentrations being observed near wharves, docks, and shipyards. In sediments, DBT and MBT predominate as a result of degradation processes, and the progression from TBT to DBT to MBT becomes more apparent with increasing distance from these foci (Gomez-Ariza *et al.*, 1995). Coastal sediments from the German areas of the North Sea and the Baltic Sea contained up to 7100 μg TBT $kg^{-1}$ wet weight and 84 000 μg TBT $kg^{-1}$ wet weight, respectively (Kalbfus *et al.*, 1996). Total butyltin concentrations in surface sediments from the Polish port of Gdynia were high compared to other coastal areas, ranging between 1800 and 2900 μg $kg^{-1}$ dry weight (Kannan and Falandysz, 1997). North Sea sediments at a

**Table A3.2.** Concentrations of butyltin species in sediments.

| Location | Year | TBT (ng $g^{-1}$ dw) | DBT (ng $g^{-1}$ dw) | MBT (ng $g^{-1}$ dw) | Reference |
|---|---|---|---|---|---|
| Baltic Sea | 1994/1995 | < 10 to 84 0000 | < 10 to 44 500 | < 10 to 53 000 | Kalbfus *et al.** (1996) |
| North Sea | 1994/1995 | 32 to 7 100 | 30 to 2 400 | 25 to 2 200 | Kalbfus *et al.** (1996) |
| North Sea, 1–70 km NW Terschelling | 1996 | 3 to 13 | 6 to 14 | 4 to 11 | RIKZ**(1999; preliminary data) |
| North Sea, 100–235 km NW Terschelling | 1996 | < 1 to 3 | < 1 to 3 | < 1 to 12 | RIKZ**(1999; preliminary data) |
| North Sea, 275 km NW Terschelling | 1996 | 27 | 21 | 17 | RIKZ**(1999; preliminary data) |
| North Sea, 2–50 km off Noordwijk | 1996 | 10 to 16 | 5 to 11 | 8 to 28 | RIKZ**(1999; preliminary data) |
| Dumping location Loswal N | 1996 | 17 | 5 | 11 | Ariese *et al.* (1997) |

*cited as wet weight **fraction < 63 μm

new disposal site for dredged harbour sediments off the Dutch coast contained on average 18 µg TBT $kg^{-1}$ dry weight in the fraction below 63 µm (Ariese *et al.*, 1997). Average levels of DBT and MBT in these sediments were 11 µg TBT $kg^{-1}$ dry weight and 25 µg TBT $kg^{-1}$ dry weight, respectively.

In order to more easily compare concentrations of organotin residues in estuarine and marine sediments from multiple studies in the future, it is strongly recommended that organotin concentrations be expressed as tin, preferentially in the fine fraction (e.g., < 63 µm) and on a dry weight basis. In any case, the basis of the concentration values presented should be clearly expressed.

As far as the persistence of tributyltin in the marine environment is concerned, many studies have shown that biological degradation is the most important factor limiting its persistence. TBT half-lives in aerobic surficial sediments in laboratory experiments ranged from 360 days to 775 days, but in anaerobic sediments no degradation was observed, and the half-life seemed to be of the order of tens of years (Dowson *et al.*, 1996). In Chinhae Bay estuarine sediment cores, DBT was found to be more persistent than TBT, with half-lives of 11.6 years and 6.9 years, respectively (Hwang *et al.*, 1999).

The disposal of TBT-contaminated sediments following dredging operations in harbours and navigation channels therefore poses particular difficulties for coastal waters, because butyltin compounds may be re-released to the water column during settling of the dredged material to the seabed, and due to the high sediment concentrations which may result at disposal sites and the potential this presents for more widespread effects on bottom fauna.

## 6 CONCLUSIONS

While reduced TBT concentrations have been observed in coastal areas frequented by small craft, this has not been the case for sites affected by inputs from large vessels (CEFIC, 1994). Similarly, although some affected dogwhelk populations have recovered since the partial bans on TBT were enacted, effects can still be observed in populations close to harbours receiving large vessels. No recovery studies have been mounted for whelks, in fact, the effect of TBT on these animals has only been investigated in one area of the North Sea. Recent studies along major shipping routes, however, indicate that water concentrations of TBT in the vicinity of large vessel traffic are still elevated relative to those in the open sea. The cessation in the decline of TBT concentrations in water in many areas, its continued presence in, and potential impact on, distant, non-target organisms, and the persistence of butyltins in sediments and the associated problems for disposal of dredgings, all provide support for an extension to the partial ban on TBT to include its use in antifouling paints on vessels of greater than 25 m in length. Such an extension was recommended by the International Maritime Organization (IMO) at the 42nd meeting of its Marine Environment Protection Committee (MEPC) in London in November 1998. Under this proposal, the application of TBT-based antifouling paints to vessels greater than 25 m in length would be prohibited from 2003, and such formulations would have to be removed from use altogether by 2008. This proposal needs further approval and ratification within IMO before it can enter into force, and the studies summarized in this review note provide additional support for this initiative.

## 7 ACKNOWLEDGEMENT

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ANNEX 4

# POLYBROMINATED BIPHENYLS AND DIPHENYLETHERS


## ABSTRACT

Polybrominated biphenyls (PBBs) and polybrominated diphenylethers (PBDEs), also called brominated diphenyl oxides, are produced and used as flame retardants. Theoretically, there are 209 different congeners of both PBBs and PBDEs. These congeners have specific chemical and physical properties, which lead to different biological and toxicological effects. Most studies have been based on commercial mixtures of brominated flame retardants, which complicates the pursuit of unambiguous data and insights. Only adequate quantification of individual congeners will allow comparative environmental and toxicological studies.

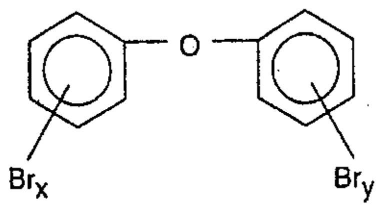
Most of the PBB and PBDE congeners found in commercial flame retardants are persistent, lipophilic and bioaccumulating, which represents a potential threat to both human life and the environment. PBDE concentrations in environmental samples are generally higher than those of PBBs, which seems to be related to the production figures of these compounds. PBB production was strongly reduced after an accident in which PBBs were mixed with cattle feed. In contrast with PBBs, PBDE production is increasing. Currently, most of the PBDE production is based on decaBDE. However, environmental patterns mainly consist of tetra- and pentaBDEs. Possibly, lower brominated mixtures have been used in higher quantities than have been reported or degradation of the relatively unstable decaBDE may take place, but other explanations cannot be excluded. The acute toxicity of PBBs and PBDEs is relatively low, but long-term effects on the balance of endocrine systems of animals and humans seem to pose the most serious risk. Because of the potential toxic effects found, a substitution of these flame retardants with environmentally friendly alternatives may be worth considering.


## 1 INTRODUCTION

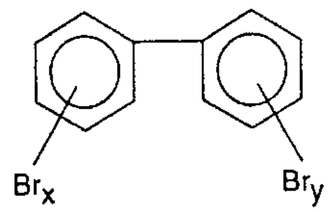
Polybrominated biphenyls (PBBs) and polybrominated diphenylethers (PBDEs) are aromatic hydrocarbons that are used as flame retardants. Flame retardants are chemicals that are added to polymers used in different materials, such as electrical and electronic equipment, paint, textiles (particularly in office buildings), and in cars and aircrafts to prevent them from catching on fire (Sellström, 1996). The use of flame retardants has increased due to stricter fire regulations in many countries and an increased use of plastic materials and synthetic fibres (Sellström, 1996). PBBs are formed by substituting hydrogen by bromine (WHO, 1994a). Instead of biphenyl, diphenylether is used in the bromination to PBDEs (WHO, 1994b).

The general chemical formulas of PBBs and PBDEs (Figure A4.1) show that PBBs and PBDEs have a large number of possible congeners, depending on the number and position of the bromine atoms on the two phenyl rings. Theoretically, 209 congeners of each chemical are possible. A systematic numbering system, developed by Ballschmiter *et al.* (1992) for polychlorinated biphenyl (PCB) congeners, has been adopted for the corresponding PBB and PBDE congeners (Pijnenburg *et al.*, 1995).

**Figure A4.1.** Basic structural formulae of PBDEs and PBBs.



PBDE (x=1-5, y=0-5)



PBB (x=1-5, y=0-5)

PBBs, manufactured in the early 1970s for commercial use, consist mainly of hexa-, octa-, nona-, and decabromobiphenyl. They were developed as flame retardants due to their ability to meet flame resistance performance requirements and their economic feasibility, as well as the fact that they have little effect on the flexibility of the base compounds. PBBs came to the attention of the public in 1974 when it was discovered that about 1000 pounds had been accidentally substituted for magnesium oxide as an additive in cattle feed in Michigan in 1973. After this, the production of PBBs has decreased (WHO, 1994a). However, decabromobiphenyl (DeBB) and possibly some other PBB mixtures are still produced commercially, although alternative chemicals have been introduced to replace them as flame retardants, in particular PBDEs. Products based on penta-, octa-, and decabromodiphenylether are the only commercially interesting PBDEs (WHO, 1994b). The production of PBDEs has increased since the end of the 1970s (WHO, 1994b). PBDEs are currently used in the housing and back covers of colour televisions and personal computers, in electronic parts of the same instruments, in seats of cars and busses, and in textiles (Pijnenburg *et al.*, 1995). PentaBDEs are mainly used in textiles and polyurethane foams, whereas decaBDE is used both in textiles and in many other kinds of synthetic plastics such as polyester used for electronic circuit boards (Sellström, 1996).

Like other organohalogen compounds such as PCBs and DDT, PBBs and PBDEs are lipophilic and persistent (WHO, 1994a, 1994b). The high resistance towards acids, bases, heat, light, reduction and oxidation is disadvantageous when these compounds are discharged into the environment, where they persist for a long time. In addition, toxic compounds, polybrominated dibenzofurans (PBDFs) and dibenzodioxins (PBDDs), may be formed when these flame retardants are heated (Pijnenburg *et al.*, 1995).

Chlorinated compounds such as PCBs (in dielectric fluids) and DDT (used as a pesticide) were found at high concentrations in living organisms in the late 1960s. These chemicals were shown to be a potential risk to various types of organisms. Since then, many countries have banned or restricted the use of these chemicals and the environmental levels have decreased (Pomerantz *et al.*, 1978). While these organochlorine compounds were banned, PBBs and PBDEs were ignored. Apart from a European ban on hexaBB production and a voluntary reduction of the PBB production in the USA, no ban has been enacted, and the production and use of brominated flame retardants have increased (Shelley, 1993). Taking into account the large worldwide production and application of PBBs and PBDEs and their persistence, it is envisaged that a large part of the total production will eventually reach the environment, including the marine environment (de Boer *et al.*, 1998b). There, the PBBs and PBDEs will bioaccumulate because of their lipophilicity and their resistance to degradation processes (Pijnenburg *et al.*, 1995). PBDEs and PBBs are considered to be a potential threat to human health, particularly through fish consumption (de Boer and Dao, 1993).

## 2 CHEMICAL AND PHYSICAL PROPERTIES

In this section, the chemical and physical properties, production and use, analytical methods, environmental fate and occurrence, and the toxicology of PBBs and PBDEs are discussed.

### 2.1 PBBs

PBBs are manufactured using a Friedel-Crafts type reaction in which biphenyl reacts with bromine in (or without) an organic solvent, using aluminium chloride, aluminium bromide, or iron as a catalyst (Brinkman and de Kok, 1980). During the production of technical-grade DeBB (Adine 0102), biphenyl is directly brominated in a large excess of bromine, used as reactant and solvent in the presence of a Lewis acid catalyst (aluminium type). DeBB is further purified by destillating the excess bromine in the presence of a brominated solvent (WHO, 1994a). The composition of the manufactured PBBs is given in Table A4.1 PBBs are not known as natural products. Of the 209 possible congeners, 101 individual PBB congeners are listed in the Chemical Abstracts Service (CAS) register at this time (WHO, 1994a).

In general, PBBs show an unusual chemical stability and resistance to acids, bases, heat, reduction and oxidation. PBBs are chemically comparable to PCBs. However, chlorine atoms have a stronger association to polybiphenyl than bromine atoms (WHO, 1994a). Unlike PCBs, the reactivity of PBBs has not been well studied and documented in the literature ((Pomerantz *et al.*, 1978). Like PCBs, their chemical stability is dependent, in part, on the degree of bromination and the specific substitution patterns (Safe, 1984).

Some chemical and physical data on commercial PBB mixtures are given in Table A4.2a. The chemical and physical properties depend on the PBB compound, and differ for each congener (WHO, 1994a).

PBBs are solid compounds with a low vapour pressure. The volatility of PBBs has a wide range and is lower than the volatility of the corresponding PCBs (Pijnenburg *et al.*, 1995). PBBs are almost insoluble in water, and the solubility decreases with increasing bromination (WHO, 1994a). Brominated compounds have a lower solubility in water than the corresponding chlorinated compounds (Pijnenburg *et al.*, 1995). Table A4.2a shows the variance in the solubility of commercial PBBs in water from different sources and qualities. Determinations of water solubility of these very hydrophobic compounds are difficult to perform. Adsorption effects on particles may influence the results. PBBs were found to be 200 times more soluble in landfill leachate than in distilled water. PBBs are soluble in various organic solvents; their solubility decreases steeply with increasing bromine content (WHO, 1994a). Most PBBs have a log $K_{ow} > 7$, and are therefore regarded as superlipophilic compounds (Prins and Meyer, 1996).

### 2.2 PBDEs

Most preparation methods of PBDEs reported are patents describing the bromination of diphenylether in the presence of a catalyst (Sellström, 1996). This results in products containing mixtures of brominated diphenylethers (Table A4.3). PBDEs have not been reported to occur naturally in the environment, but the related polybrominated phenoxy phenols have been found in several marine organisms, e.g., in *Dysidea herbacea*, *Dysidea chlorea*, and *Phyllospongio foliascents* (WHO, 1994b). Vionov *et al.* (1991) showed that the bacteria *Vibrio* sp. associated with the sponge *Dysidea* sp. is capable of producing brominated diphenylethers.

**Table A4.1.** Composition of commercial PBB mixtures (WHO, 1994a).

| **PBB mixture (manufacturer)** | **Weight of bromine (%)** | **Weight of different homologous groups** | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | | $Br_{10}$ | $Br_9$ | $Br_8$ | $Br_7$ | $Br_6$ | $Br_5$ | $Br_4$ |
| **Hexabromobiphenyl** | | | | | | | | |
| FM BP-6 (Michigan Chemical) | 75 | | | | 13.8 | 62.8 | 10.6 | 2 |
| FM [lot RP-158 (1971)] | | | | | 12.5 | 72.5 | 9 | 4 |
| FM [Lot 6244A (1974)] | | | | | 13 | 77.5 | 5 | 4.5 |
| FM BP-6 | | | | | | 90 | 10 | |
| FM BP-6 | | | | 1 | 18 | 73 | 8 | |
| FM BP-6 | | | | | 33 | 63 | 4 | |
| FM BP-6 | | | | | 7.7 | 74.5 | 5.6 | |
| FM BP-6 | | | | | 24.5 | 79 | 6 | |
| 2,2',4,4',6,6' (RFR) | | | | | 12 | 84 | 1 | |
| 2,2',4,4',6,6' (Aldrich) | | | | 2 | 24 | 70 | 4 | |
| 'Hexabromobiphenyl' (RFR) | | | | | 25 | 67 | 4 | |
| | | | | | (12–25) | (60–80) | (1–11) | (2–5) |
| **Octanonabromobiphenyl** | | | | | | | | |
| Bromkal 80-9D (Kalk) | 81–82.5 | 9 | 65 | 25 | 1 | | | |
| Bromkal 80 | | | | 72 | 27 | 1 | | |
| XN-1902 (Dow Chemical) | 82 | 6 | 47 | 45 | 2 | | | |
| XN-1902 (Dow Chemical) | | 2 | 34 | 57 | 7 | | | |
| Lot 102-7-72 (Dow Chemical) | | 6 | 60 | 33 | 1 | | | |
| 'Octabromobiphenyl' (RFR) | | 4 | 54 | 38 | 2 | | | |
| 2,2',3,3',5,5',6,6' (RFR) | | 1 | 28 | 46 | 23 | 2 | | |
| FR 250 13A (Dow Chemical) | | 8 | 49 | 31 | 1 | | | |
| **Decabromobiphenyl** | | | | | | | | |
| HFO 101 (Hexcel) | 84 | 96 | 2 | | | | | |
| Adine 0102 (Ugine Kuhlmann) | 83–85 | 96 | 4 | | | | | |
| Adine 0102 (Ugine Kuhlmann) | | 96.8 | 2.9 | 0.3 | | | | |
| Decabromobiphenyl (RFR) | | 71 | 11 | 7 | 4 | 4 | | |
| DBB: Flammex B 10 (Berk) | | 96.8 | 2.9 | 0.3 | | | | |

Because of the presence of an oxygen atom, there is less similarity in the molecular structure between PBDEs and PCBs than between PBBs and PCBs (WHO, 1994b). The commercial PBDEs are rather stable compounds with boiling points ranging between 310 °C and 425 °C (WHO, 1994b), and with low vapour pressures (Table A4.2b) (Sellström, 1996; WHO, 1994b). The volatility of PBDEs is low and their solubility in water is very low, especially that of higher brominated diphenylethers. It has been concluded that higher brominated compounds are more persistent than lower brominated compounds. PBDEs are soluble in toluene. The commercial PBDEs are lipophilic substances for which the log $K_{ow}$ increases with increasing bromine content (WHO, 1994b).

## 3 PRODUCTION AND USE

PBBs and PBDEs belong to a group of brominated organic compounds which are used as flame retardants. Flame retardants are valued for their ability to inhibit combustion in plastics, textiles, electric, and other materials. There are different groups of flame retardants: inorganic and organic chemicals. Usually they are divided into reactive and additive flame retardants.

Reactive flame retardants have the same functional groups as the monomer with which they react. They are covalently bound to the polymer and are therefore less likely to leach to the environment. Reactive-type flame retardants offer advantages such as polymer strength

**Table A4.2a.** Chemical and physical data on commercial PBB mixtures (WHO, 1994a, 1994b; Pijnenburg *et al.*, 1995).

| | **HxBB ($C_{12}H_4Br_6$)** | **OcBB ($C_{12}H_2Br_8$)** | **NoBB ($C_{12}H_1Br_9$)** | **DeBB ($C_{12}Br_{10}$)** |
|---|---|---|---|---|
| Relative molecular mass* | 627.4 | 785.2 | 864.1 | 943.0 |
| Melting point (°C) | 124–248 | 200–250 | 220–290 | 380–386 |
| | | | 360–380 | |
| | | | 385 | |
| Decomposition point (°C) | 300–400 | 435 | 435 | 395>400 |
| Volatility (% weight loss) | | <1 % at 250 °C | 1–2 % at 300 °C | <5 % at 341 °C |
| | | <10 % at 330 °C | | <10 % at 363 °C |
| | | <50 % at 350 °C | | <25 % at 388 °C |
| Vapour pressure (Pa) | 25 °C; 0.000007 | | | <0.0000006 (temperature not given) |
| | 90 °C; 0.01 | | | |
| | 140 °C; 1 | | | |
| | 222 °C; 100 | | | |
| Solubility $H_2O$ | 11 | 30–40 | insoluble | <30 |
| (µg/l; 25 °C) | 610 | | | |
| destilled | 0.32 | | | |
| deionized | 0.06 | | | |
| pure BB 153 | 30 | | | |
| soluble in: (g kg$^{-1}$ solvent; at 28 °C) | carbontetrachloride; 300 chloroform; 400 | petroleum ether; 18 benzene; 81 | insoluble | carbontetrachloride; 10 |
| | benzene; 750 | | | |
| | toluene; 970 | | | |
| | dioxane; 1150 | | | |
| Log $K_{ow}$* | 7.20 | | | 8.58 |

**Table A4.2b.** Chemical and physical data on commercial PBDE mixtures (WHO, 1994a, 1994b; Pijnenburg *et al.*, 1995).

| | **TeBDE ($C_{12}H_4Br_4O$)** | **PeBDE ($C_{12}H_2Br_6O$)** | **OcBDE ($C_{12}H_2Br_8O$)** | **DeBDE ($C_{12}Br_{10}O$)** |
|---|---|---|---|---|
| Relative molecular mass* | 485.82 | 564.75 | 801.47 | 959.22 |
| Melting point (°C) | | –7 – –3 | 200 | 290–306 |
| | | (boiling 300) | 79–89 | |
| | | | 75–125 | |
| | | | 170–220 | |
| Decomposition Point (°C) | | >200 | | >320 |
| | | | | >400 |
| | | | | >425 |
| Volatility (% weight loss) | | | | 1 % at 319 °C |
| | | | | 5 % at 353 °C |
| | | | | 10 % at 370 °C |
| | | | | 50 % at 414 °C |
| | | | | 90 % at 436 °C |
| Vapour Pressure (mm Hg) | | 22 °C; 9.3 | 25 °C <$10^{-7}$ | 20 °C <$10^{-7}$ |
| | | | | 250 °C <1 |
| | | | | 278 °C; 2.03 |
| | | | | 306 °C; 5.03 |
| Solubility $H_2O$ (at 25 °C) | | $9 \times 10^{-7}$ mg l$^{-1}$ at 20°C | <1 g l$^{-1}$ | 20–30 µg l$^{-1}$ |
| soluble in: | | methanol; 10 | toluene, 190 (353) | *o*-xylene 8.7 |
| (g kg$^{-1}$ solvent; at 25 °C) | | (and other organic solvents) | benzene; 200 styrene; 250 | |
| Log $K_{ow}$* | 5.87–6.16 | 6.64–6.97 | 8.35–8.90 | 9.97 |

* data for BDE congeners

**Table A4.3.** Composition of commercial PBDEs (WHO, 1994b).

| Product | Composition | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | PBDE[a] | TrBDE | TeBDE | PeBDE | HxBDE | HpBDE | OBDE | NBDE | DeBDE |
| DeBDE | | | | | | | | 0.3–3 % | 97–98 % |
| OBDE | | | | | 10–12 % | 43–44 % | 31–35 % | 9–11 % | 0–1 % |
| PeBDE | | 0–1 % | 24–38 % | 50–62 % | 4–8 % | | | | |
| PBDE[b] | 7.6 % | – | 41–41 % | 44.4–45 % | 6–7 % | | | | |

[a]Unknown structure; [b]No longer commercially produced; analysis of one single congener.

**Table A4.4.** Commercial production of PBBs in the USA, 1970–1976 (Di Carlo *et al.*, 1978).

| | Estimated production (in 1000 kg) | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| **Product** | **1970** | **1971** | **1972** | **1973** | **1974** | **1975** | **1976** | **1970–1976** |
| Hexabromobiphenyl | 9.5 | 84.2 | 1011 | 1770 | 2221 | 0 | 0 | 5369 |
| Octabromobiphenyl and decabromobiphenyl | 14.1 | 14.1 | 14.6 | 163 | 48 | 77.3 | 366 | 702 |
| Total PBBs | 23.6 | 98.3 | 1025.6 | 1922 | 2269 | 77.3 | 366 | 6071 |

permanency and solvent resistance. A disadvantage is that they are polymer specific (Hairston, 1995).

Additive flame retardants are not chemically incorporated into the polymer molecule. The additives are only mixed with or dissolved in the material and can therefore migrate out of the product throughout its entire lifetime (Sellström, 1996).

What all flame retardants have in common is that they start to decompose when heated. A critical factor in the selection of a flame retardant is therefore its thermal stability with respect to that of the polymer. The ideal situation is when the flame retardant decomposes at about 50 % below the combustion temperature of the polymer. This is the case with most organic bromine compounds and most synthetic polymers (Sellström, 1996). In addition, brominated flame retardants are economically feasible, and they have little effect on the flexibility of the base compounds (Mumma and Wallace, 1975). Because of the advantages mentioned above, the commercial use of brominated compounds is attractive.

### 3.1 PBBs

The commercially used PBB mixtures consist mainly of hexaBB, octaBB, nonaBB, and decaBB (Table A4.1). Commercially manufactured PBBs are primarily processed as flame retardants. Further potential uses of PBBs are: in the synthesis of biphenylesters or in a modified Wurtz-Fittig synthesis, in light-sensitive compositions to act as colour activators, as relative molecular mass control agents for polybutadiene, as wood preservatives, as voltage stabilizing agents in electrical insulation, and as functional fluids such as dielectric media (Neufeld *et al.*, 1977).

PBBs were introduced as flame retardants in the early 1970s. In the USA, the production of hexaBB ceased as a result of the Michigan disaster (Table A4.4). OctaBB and decaBB were produced until 1979 (WHO, 1994a). There was no import to the USA of any PBB mixtures. Since 1975/1976, all PBBs manufactured in the USA have been exported, mainly to Europe (Brinkman and de Kok, 1980). In Japan some PBBs were imported until 1978, but there was no production. A mixture of highly brominated PBBs (Bromkal 80-9D) was produced in Germany until mid-1985 (WHO, 1994a). The production of DeBB in Great Britain was discontinued in 1977 (Neufeld *et al.*, 1977). The French company Atochem is currently producing technical grade DeBB (Adine 0102) in quantities of a few hundred thousand kg per year. It is marketed in France, Great Britain, Spain, and the Netherlands (Anon., 1990). In the Netherlands more than 200 tonnes DeBB per year (1989) were used (UBA, 1989). The use of PBB mixtures in European textiles is not allowed according to EU Directive 83/264 (Bjerregaard, 1998).

Most research on PBBs has been carried out on Fire Master BP-6 and FF-1, which were involved in the Michigan disaster. The PBB composition of Fire Master changes from batch to batch (Table A4.1), but also mixed bromochlorobiphenyls and polybrominated naphthalenes have been observed as minor components (WHO, 1994a). Approximately twenty compounds other than PBBs were tentatively identified in Fire Master (Hass *et al.*, 1978). An extensive study was performed on a large number of batches of Fire Master, analysed for the toxic compounds PBDDs and PBDFs. These compounds were found in only one sample of Adine 0102 (WHO, 1994a).

## 3.2 PBDEs

Products based on penta-, octa-, and decaBDE are of commercial interest (see Table A4.3). PBDEs are mainly used as a flame retardant. There are nine manufacturers who currently produce PBDEs. They are listed in Table A4.5.

**Table A4.5.** Manufacturers currently producing PBDE compounds.

| | |
|---|---|
| Dead Sea Bromines and Eurobrome | Netherlands |
| Atochem | France |
| Great Lakes Chemical Ltd | Great Britain |
| *Great Lakes Chemical Corporation* | USA |
| Albemarle | USA |
| Tosoh | Japan |
| Matsunaga | Japan |
| Nippo | Japan |

The global production of decaBDE (DeBDE) is approximately 40 000 tonnes per year (WHO, 1994b; Sellström, 1996; Spiegelstein, 1998). Penta-, octa-, and decaBDE are currently being evaluated by the European Commission according to EU Directive 793/93 (Bjerregaard, 1998).

## 4 CONSUMPTION

Due to more stringent fire regulations in many countries and the increased use of plastic materials and synthetic fibres, the use of flame retardants has increased. In 1992, 600 000 tonnes of flame retardants were used worldwide; 150 000 tonnes were brominated compounds. Fifty thousand tonnes of these were the reactive flame retardant with tetrabromylbisphenyl-a (TBBPA) and its derivatives and 40 000 tonnes were PBDEs (Sellström, 1996).

The annual global consumption of PBDEs is 40 000 tonnes (30 000 tonnes DeBDE, 6000 tonnes OcBDE and 4000 tonnes PeBDE) (Sellström, 1996; WHO, 1994b). Data on the usage of PBDEs (WHO, 1994b) are shown in the following table:

| | |
|---|---|
| Germany | 3000–5000 tonnes year$^{-1}$ (1991) |
| Sweden | 1400–2000 tonnes year$^{-1}$ (1991) |
| Sweden | 400 tonnes year$^{-1}$ (1991) (Sellström, 1996) |
| Netherlands | 3300–3700 tonnes year$^{-1}$ (1992) |
| Great Britain | 2000 tonnes year$^{-1}$ (1993) |

TeBDEs were not used in Japan after 1990. Inorganic flame retardants (including aluminium trihydrate (ATH)) were mainly used, but there has been an increase in the use of brominated organic flame retardants (Sellström, 1996).

In the USA, brominated flame retardants belong to the most widely used additive flame retardants (Table A4.6). It is expected that in the USA the demand for additive products will increase by about 5.3 % per year, to 476 700 tonnes in 1998. Despite their specificity, the demand for reactive products is expected to increase by 4.6 % per year to 68 100 tonnes in 1998, as new products are introduced on the market. It is expected that the brominated flame retardant consumption will increase up to over 50 000 tonnes in the USA (Hairston, 1995).

**Table A4.6.** The flame retardant demand in 1993 and the expected flame retardant demand in 1998 in the USA, in tonnes (Hairston, 1995).

| Product | 1993 | 1998 |
|---|---|---|
| Additive flame retardants (total) | 367 740 | 476 700 |
| Aluminatrihydrate | 196 128 | 256 510 |
| Phosphorus compounds | 44 038 | 57 658 |
| Bromine compounds | 39 952 | 50 848 |
| Antimony oxide | 28 148 | 35 412 |
| Chlorinated compounds | 24 970 | 29 964 |
| Boron compounds | 7 264 | 9 080 |
| Other additives | 27 240 | 37 228 |
| Reactive flame retardants (total) | 54 480 | 68 100 |
| Epoxy intermediates | 16 344 | 19 976 |
| Polyester intermediates | 12 258 | 14 528 |
| Urethane intermadiates | 9 080 | 10 896 |
| Polycarbonate intermediates | 7 264 | 9 988 |
| Other intermediates | 9 534 | 12 712 |
| **TOTAL Demand** | **422 220** | **544 800** |
| Percent Additive | 87.1 % | 87.5 % |
| Percent Reactive | 12.9 % | 12.5 % |

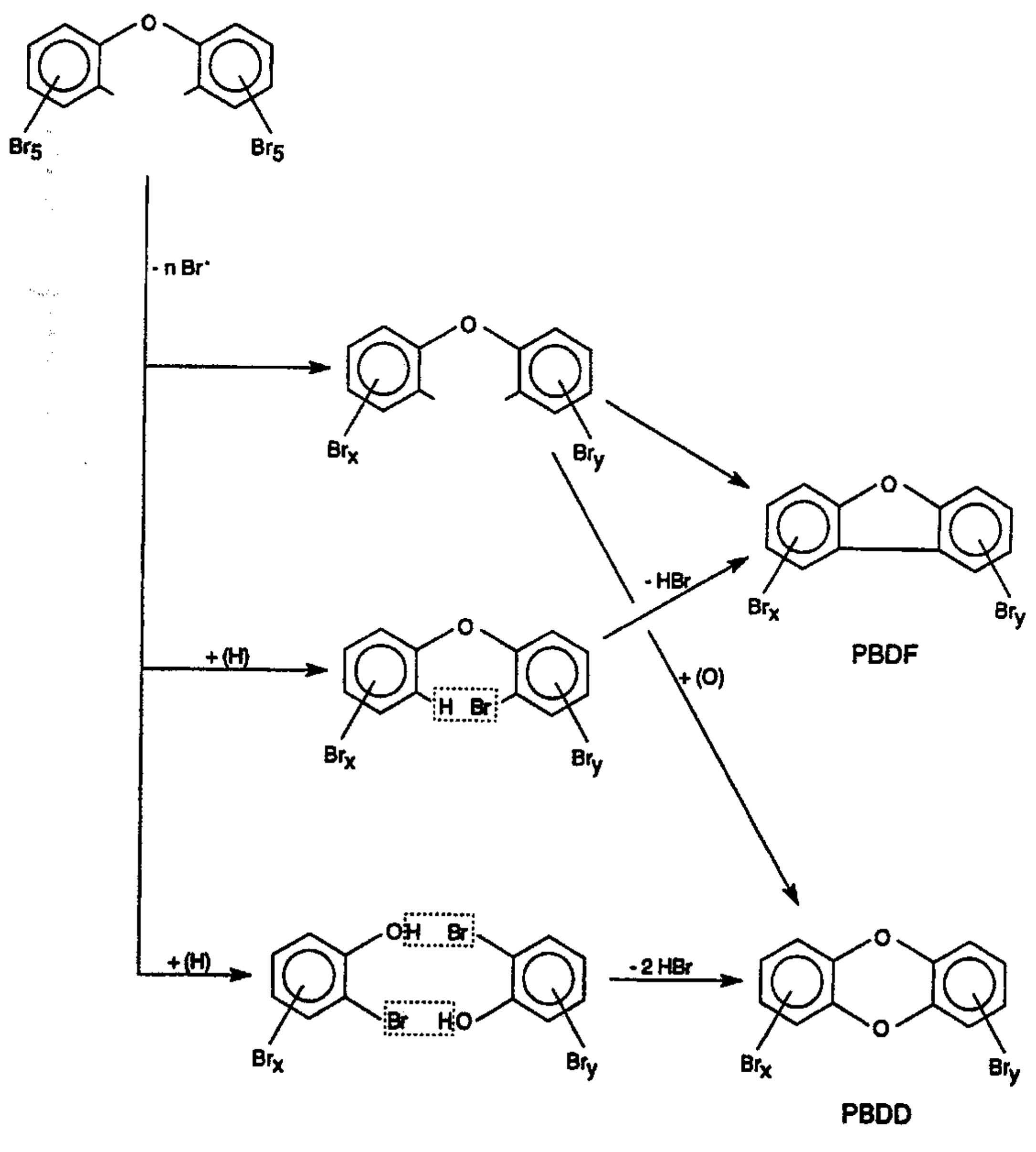
Compared with data on the worldwide use in 1992, the demand for flame retardants in the USA in 1993 was extremely high. However, only a small fraction of this consisted of brominated compounds. The use of brominated compounds as flame retardants in Japan in 1993 was higher, both relatively and in absolute amounts. The 1998 USA consumption of brominated compounds (in absolute amounts) was expected to reach the same level as that of Japan in 1994. Unfortunately, accurate data on flame retardant demands in Europe are not available. The European consumption of brominated compounds is estimated to be at a similar level as in Japan and the USA.

## 5 COMBUSTION AND RECYCLING OF PBBs AND PBDEs

A disadvantage of brominated flame retardants is that PBDDs and PBDFs may be formed during combustion. Little is known about the toxicity of PBDDs and PBDFs. Toxicities are estimated to be in the same order as those of polychlorinated dibenzodioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) (WHO, 1994a, 1994b).

Debromination reactions of higher brominated flame retardants lead to lower brominated PBDF and PBDD congeners (WHO, 1994b). On pyrolysis PBDEs produce larger amounts of dioxins and furans than PBBs, and in this respect PBDEs are more toxic than PBBs. The most likely mechanism is that, with the oxidation of PBDEs, PBDFs and PBDDs are formed in intramolecular cyclization reactions involving the attack by oxygen on the diphenylether system (Figure A4.2) (Bieniek *et al.*, 1989). Most of the reports have indicated that maximum production of PBDFs and/or PBDDs was observed at temperatures of 400–800 °C, depending on the type of brominated flame retardant, and that the 2,3,7,8-substituted compounds were seen only in very low concentrations (WHO, 1994b). At 600 °C, 2,3,7,8-TeBDD and TeBDF in concentrations of 0.01–7 mg $kg^{-1}$ and 0.01–6 mg $kg^{-1}$, respectively, are formed from plastics containing DeBDE or PBDE as flame retardant. With increasing temperature, the concentration of these isomers decreases until they are no longer detectable above 800 °C (detection limit 0.01 mg $kg^{-1}$) (Lahaniatis *et al.*, 1991).

**Figure A4.2.** Possible mechanisms for the formation of PBDFs and PBDDs from DeBDE (Bieniek *et al.*, 1989).



Studies have been performed on the photodegradation of DeBDE in organic solvents and water. DeBDE was irradiated in hexane solution with UVR and sunlight (WHO, 1994b). A mixture of tri- to octaBDE congeners was detected. In addition, a large number of PBDFs containing 1–6 bromoatoms and small amounts of polybromobenzenes were formed. Photodegradation of PBDE in water does not lead to the formation of lower BDE or BDF (WHO, 1994b). Sellström *et al.* (1998b) showed a rapid degradation of DeBDE in toluene (half-life: 15 minutes) after UV radiation and in sand (half-lives: 12–37 hours) after UV and sunlight radiation. Lower PBDEs and other compounds were found after irradiation. Sellström *et al.* (1998a) reported that the photolysis of DeBDE during extraction may be prevented by the presence of coextractives from the sample. Watanabe *et al.* (1994) showed the formation of polybrominated dibenzofurans after UV radiation of DeBDE in hexane. This possible degradation of DeBDE should be taken into account when establishing conditions for the analysis of DeBDE. Watanabe *et al.* (1986) reported that DeBDE dissolved in hexane can be degraded to NoBDE, OcBDE, HpBDE, and HxBDE.

No degradation of PBBs by plants has been recorded. In contrast to plants, animals readily absorb PBBs (WHO, 1994a). Data on environmental fate (although limited to MBDE, DiBDE, DeBDE) suggest that biodegradation is not an important degradation pathway for PBDEs, but that photodegradation may play a significant role (WHO, 1994b).

Environmental studies so far indicate a possible partial degradation of the original PBDEs and PBBs to lower brominated congeners. Because the carbon-bromine bond is less stable than the carbon-chlorine bond, reductive debromination may be a degradative pathway of bromobiphenyls and this reaction may have toxicological consequences not encountered with PCBs (WHO, 1994a). *In vitro* studies with hepatic microsomes of marine mammals showed a high persistency of a number of PBB and PBDE congeners (de Boer *et al.*, 1998c).

## 9 ENVIRONMENTAL LEVELS AND HUMAN EXPOSURE

Some typical PBDE and PBB concentrations in environmental samples are given in Table A4.8. Most concentrations determined to date are semi-quantitative total PBB or PBDE concentrations, but recently more congener-specific data have become available.

### 9.1 PBBs

The only report (Stratton and Whitlock, 1979) on PBB levels in the air concerns air samples taken in the vicinity of three PBB plants in the USA. Traces of HxBB (0.06–1.10 ng $m^{-3}$) were found in two samples. Depending on its source, the predominant PBB compounds detected in surface water were HxBB and DeBB (WHO, 1994a).

Many studies started after the accidental contamination in 1973 in Michigan, with Fire Master FF-1 being inadvertently substituted for magnesium oxide in the production of cattle feed. Estimates of the amount of PBBs used vary between approximately 290 kg (Fries, 1985) to 1000 kg (IARC, 1978). PBBs were mixed into feeds, and distributed widely to Michigan farmers. In addition, feeds not formulated to contain magnesium oxide became contaminated (at relatively low concentrations) due to the carry-over of PBBs from batch to batch through mixing equipment and, on farms, through the recycling of contaminated products. The mixing error was not discovered immediately, and it was almost a year before analysis indicated that a compound of PBB was involved in the illness or death of farm animals. During this time, contaminated animals and their products entered the human food supply and the environment of the state of Michigan (WHO, 1994a).

Groundwater near local disposal sites was not contaminated by PBBs (Shah, 1991). Soils from PBB industrial sites (2000 mg $kg^{-1}$ dry weight, Fire Master plant) have in general been more heavily contaminated than Michigan soils (371 μg $kg^{-1}$ dry weight) (Fries, 1985; Jacobs *et al.*, 1978). Contamination of animal feed or foods by PBBs has been reported only in connection with the Michigan PBB incident (WHO, 1994a).

High levels of nonaBB and octaBB (in addition to PBDE) were present in fish from German rivers (WHO, 1994a). However, HxBBs were predominant in fish from the North Sea and the Baltic Sea (WHO, 1994a; Pijnenburg *et al.*, 1995). In all samples from the Baltic Sea, 3,3′,4,4′,5,5′-HxBB was found at relatively high concentrations (maximum concentration: 36 μg $kg^{-1}$ fat), but it was not detected in the North Sea or in rivers (Jansson *et al.*, 1991). The concentrations of other HxBBs are usually higher in fish from the Baltic Sea than in fish from the North Sea. Concentrations of BB-153 (2,2′,4,4′,5,5′-hexaBB) determined in marine fish ranged from 0.2–2.4 μg $kg^{-1}$ lipid (Baltic fish), and in seals from 0.4 μg $kg^{-1}$ lipid (Northern Ice Sea) to 26 μg $kg^{-1}$ lipid (Baltic Sea) (Jansson *et al.*, 1991, 1993). The congener pattern found in fish is quite different from that found in commercial products. Many of the major peaks could well be the result of photochemical debromination of DeBB, but this has not been confirmed (WHO, 1994a). Long-range transport has not been proved, but the presence of these compounds in Arctic seal tissue samples indicates a wide geographical distribution (WHO, 1994a).

For most human populations, direct data on exposure to PBBs from various sources have not been documented. Occupational exposure was found in employees in chemical plants in the USA (skin contact and inhalation) and in farm workers (skin contact, inhalation, and contaminated food). Median serum and adipose tissue PBB levels were higher among chemical workers (WHO, 1994a).

**Table A4.8.** A selection of typical PBB and PBDE concentrations in the aquatic environment.

| Matrix | Location | PBB (μg kg$^{-1}$) | PBDE (μg kg$^{-1}$) | Reference |
|---|---|---|---|---|
| Sediment | Osaka May | | 11–30 dw, tetra | Watanabe and Tatsakawa, 1990 |
| Sediment | Japanese rivers | | 33–375 dw, deca | Watanabe and Tatsakawa, 1990 |
| Sediment | River Tees, UK | | 1348 dw, total | Nylund *et al.*, 1992 |
| Sediment | Mersey, UK | 0.33 dw, deca<br><0.01 dw, hexa | 1700 dw, deca<br>2.2 dw, tetra | Allchin *et al.*, 1999 |
| Sediment | Western Scheldt, Netherlands | 0.84 dw, deca<br>0.024 dw, hexa | 200 dw, deca<br>0.42 dw, tetra | Allchin *et al.*, 1999 |
| Sediment | Rhine | 0.39 dw, deca<br>0.04 dw, hexa | 15.7 dw, deca<br>1.4 dw, tetra | Allchin *et al.*, 1999 |
| Sediment | Elbe, Germany | <0.1 dw, deca<br><0.01 dw, hexa | 0.83 dw, deca<br><0.17 dw, tetra | Allchin *et al.*, 1999 |
| Mussels | Osaka Bay | | 15 ww, tetra | Allchin *et al.*, 1999 |
| Cod liver | Southern North Sea | | 170 lw, tetra<br>5 lw, penta[a] | de Boer, 1989 |
| Mackerel | North Sea | 0.04 ww, hexa | 5.4 ww, tetra | de Boer *et al.*, 1998b |
| Pike | Southern Sweden | | 27 000 lw, tetra | Sellström *et al.*, 1990 |
| Herring | Baltic Sea | 0.16 lw, total | 528 lw, total | Jacobs *et al.*, 1978 |
| Herring | Skagerrak | 0.27 lw, total | 735 lw, total | Jacobs *et al.*, 1978 |
| Flounder liver | Humber, UK | | 217 ww, tetra<br>22 ww, penta | Nylund *et al.*, 1992 |
| Flounder liver | Tees Bay, UK | | 1294 ww, tetra<br>238 ww, penta | Nylund *et al.*, 1992 |
| Salmon | Baltic Sea | | 167 lw, tetra<br>96 lw, penta<br>12.7 lw, hexa | Haglund *et al.*, 1997 |
| Harbour seal | North Sea | 13–61 ww, hexa<br><1 ww, deca | 280–1200 ww, tetra<br>140–270 ww, penta<br><10 ww, deca | de Boer *et al.*, 1998b |
| Grey seal | Baltic Sea | | 308 lw, tetra<br>101 lw, penta<br>38 lw, hexa | Haglund *et al.*, 1997 |
| Whitebeaked dolphin | North Sea | 13 ww, hexa<br>8.3 ww, penta<br><0.9 ww, deca | 5500 ww, tetra<br>2200 ww, penta<br><10 ww, deca | de Boer *et al.*, 1998b |
| Sperm whale | Atlantic Ocean | 1.1–1.9 ww, tetra<br>0.4–0.9 ww, penta<br><0.5 ww, deca | 58–95 ww, tetra<br>17–40 ww, penta<br><5 ww, deca | de Boer *et al.*, 1998b |
| Pilot whale | Faroe Islands | | 435–1850 lw, tetra<br>265–940 lw, penta<br>140–370 lw, hexa | Lindström *et al.*, 1999 |
| Cormorant liver | Rhine delta | | 28 000 ww, total | de Boer, 1990 |
| Sea eagle | Baltic Sea | 280 lw, total | 350 lw, total | Jansson *et al.*, 1987 |
| Guillemot | Northern Ice Sea | 50–130 lw, total | | Jacobs *et al.*, 1978;<br>Jansson *et al.*, 1987 |

dw = dry weight; lw = lipid weight; ww = wet weight; [a]penta: often based on two congeners.

Recently, PBBs (and also PBDEs) have been detected in cow milk and human milk in Germany (Krüger, 1988). The congener patterns in these samples differ from that found in fish. BB-153 was the most abundant component in human milk (Krüger, 1988). The relative concentration of BB-153 is higher in human milk (1.03 μg $kg^{-1}$ lipid) (Krüger, 1988) than in fish (ranged between 0.092–24 μg $kg^{-1}$ lipid) (Krüger, 1988; Jansson *et al.*, 1993). Total levels found in human samples were substantially higher than levels that were detected in cow milk (both samples, cow and human, from the same region) (Krüger, 1988). Thus, an infant of 6 kg body weight consuming human milk will have a higher intake of PBBs than an adult consuming cow milk, 0.01 μg PBB $kg^{-1}$ body weight per day and 0.00002 μg PBB $kg^{-1}$ body weight per day, respectively (WHO, 1994a).

## 9.2 PBDEs

In Japan, a large amount of PBDEs was determined in the airborne dust; DeBDE was observed as the dominant congener (83–3060 pg $m^{-3}$), while other congeners were TeBDE, PeBDE, and HxBDE (Watanabe *et al.*, 1995). These PBDEs were also present in two ash and soil samples from a recycling plant in Taiwan, in which DeBDE was the dominant congener (510–2500 μg $kg^{-1}$ ash and 260–330 μg $kg^{-1}$ soil) (Watanabe *et al.*, 1993).

Two samples of sewage sludge from the same sewage treatment plant in Gothenburg (Sweden) were analysed. One sample was a pool of subsamples taken during a period with little rain and the other was composed of subsamples during a rainy period. The levels were 25 and 21 ng $g^{-1}$ dry weight for the dry and wet periods, respectively, indicating that the primary PBDE sources to this matrix are household and industrial effluents and not washout from the atmosphere (Sellström, 1996).

Surficial sediment samples up- and downstream from a plastics industry in Sweden indicated this industry as the most likely source. The relative amounts in the analysed sewage sludge and surficial sediment samples are quite similar to the pattern for the technical PBDE product Bromkal 70-5 DE (Sellström, 1996). De Boer and Dao (1993) found a PBDE pattern in sediments that is comparable to the pattern of this technical mixture. In these sediment samples, PeBDE concentrations were higher than TeBDE concentrations. TeBDE, PeBDE, and HxBDE were found in sediments of Osaka Bay (Japan), and in seven of fifteen riverine and estuarine samples, DeBDE was found in higher concentrations (Pijnenburg *et al.*, 1995), indicating accumulation of higher brominated congeners in the sediment. Recently, DeBDE has been detected for the first time in Sweden in some sediment samples from the Viskan River and in sludge samples (Sellström, 1996; Sellström *et al.*, 1998a). The upper layer in a laminated sediment core from the Baltic Sea contained higher levels of TeBDE and PeBDE than lower layers, indicating an increasing burden of these compounds (Nylund *et al.*, 1992). Other time-trend studies of Baltic sediments showed an increasing trend in the concentrations of PBDEs between 1973 and 1990 (Sellström, 1996). PBDEs seem to have a higher absorption to the sediment than PCBs (de Boer and Dao, 1993). Allchin *et al.* (1999) found relatively high tetra-BDE concentrations in sediments of the Tees and Skerne rivers in the UK, whereas in other UK rivers tetra- and pentaBDEs were also detected. In the same study, PBDEs were found in liver and muscle tissues of several fish species from the same rivers and estuaries (Table A4.8). In a survey on PBDE and PBB concentrations in sediments from 22 estuaries in Europe, high decaBDE concentrations were found in the Mersey and Scheldt rivers (Table A4.8) (Anon., 1997). At all locations, decaBDE concentrations were higher than tetraBDE concentrations.

PBB concentrations are relatively low and are often below detection limits. Swedish results from freshwater fish studies (Table A4.8) indicate that southern Sweden may be more contaminated with PBDEs than northern Sweden (Sellström, 1996). Compared with levels found in terrestrial animals (rabbit, moose, reindeer at 0, 1.7, 0.47 ng $g^{-1}$ lipid weight, respectively) Jansson *et al.* (1993) showed that the concentrations of PBDEs are higher in aquatic organisms than in terrestrial organisms (Sellström, 1996; WHO, 1994b). Sellström (1996) and Watanabe *et al.* (1987) detected PBDEs in freshwater fish (Sweden and Japan), with TeBDE congeners dominant in the samples. Comparing these results with Bromkal 70-5 DE, the relative amount of TeBDEs is much higher (Sellström, 1996). In addition, de Boer and Dao (1993) found relatively higher concentrations of TeBDEs in biological samples. The PBDE pattern in sediments is more comparable to the pattern of Bromkal 70-5 DE than the PBDE pattern in biological samples. Relatively higher TeBDE concentrations in fish may be caused by a more rapid uptake of lower brominated compounds. Possibly, a membrane barrier exists for higher brominated compounds due to the larger size of these molecules. Consequently, in sediments relatively more higher brominated compounds may be expected (de Boer and Dao, 1993).

Noticeable concentrations of PBDEs were determined in carp of three age classes collected from the Buffalo River (New York, USA). TeBDEs accounted for 94–96 % of total PBDE concentrations (Loganathan *et al.*, 1995).

In Japan, as in Europe, TeBDEs are the major component in marine and shellfish samples (Watanabe *et al.*, 1995).

Contamination by PBDEs on a lipid weight basis is about nine times higher in herring caught in the spring than in herring caught in the autumn. This relationship has previously been shown for PCBs, DDT, and dioxins. Spring herring is caught near the breeding season and this probably affects lipid disposition and metabolism, which in turn may affect concentrations of organohalogens (Sellström, 1996).

Several fish-eating animals have been studied. Extremely high PBDE levels (up to 25 000 μg $kg^{-1}$ TeBDE and 4000 μg $kg^{-1}$ PeBDE (wet weight)) were found in liver tissue of a cormorant from the Rhine delta. Since this is based on only a single animal, further research is necessary (de Boer and Dao, 1993). PBDE levels found in an osprey from Sweden, which also feeds on freshwater fish, were high as well (160–1900 $\mu g\ kg^{-1}$ lipid) (Sellström, 1996). Baltic seals contained higher TeBDE concentrations than North Sea seals (de Boer and Dao, 1993). Time-trend studies of guillemot eggs (Stora Karslö, Baltic Sea, Sweden) indicate that the levels of PBDEs have increased since 1970 and that this increase is significant. A similar time-trend study of pike (from Lake Bolmen, Sweden) shows a similar trend. However, in guillemot eggs there are indications that the PBDE levels may have decreased during recent years (Sellström, 1996). Both guillemots and grey seals show higher concentrations of three PBDE congeners, 2,2′,4,4′-TeBDE, an unknown PeBDE, and 2,2′,4,4′,5-PeBDE, than the herring they feed on. TeBDE seems to biomagnify to the relatively highest extent (Sellström, 1996). Biomagnification is also observed in dolphins and porpoises from the southern North Sea and the Atlantic west of Ireland. Biomagnification factors between fish and investigated marine mammals are approximately 10–30 (de Boer and Dao, 1993).

Only long-range transport through air can explain the contamination in whitefish in Lake Storvindel, Sweden. PBDEs have been found in an air sample collected near this lake. In another air sample collected at the southern part of the Baltic Sea, PBDEs were detected as well (Sellström, 1996).

In Germany, cow milk was analysed and an average concentration of 3.57 $\mu g\ kg^{-1}$ fat (four samples) was found and determined as Bromkal 70-5 DE; the main component was HxBDE. PBDEs were also detected in human breast milk (Germany). The samples contained 0.6–11.1 $\mu g$ PBDE $kg^{-1}$ fat, determined as Bromkal 70-5 DE (Krüger, 1988). Uptake of TeBDEs and PeBDEs may occur in humans via the food chain, e.g., by consuming fish. Exposure may also occur through skin contact (flame retardants in polymers used in textiles) and via inhalation (WHO, 1994b).

In general, environmental PBDE concentrations are considerably higher than those of PBBs. De Boer *et al.* (1998b) found PBDE concentrations fifty-fold higher than PBB concentrations in sperm whales. This corresponds with the production of PBDEs which is still ongoing and increasing, whereas the PBB production is mainly restricted to DeBB and seems to be relatively small. One of the questions to be solved is the difference between the current PBDE production, which is mainly based on DeBDE with a relatively smaller amount of PeBDEs, and the patterns found in the environment, which show the highest concentrations of 2,2′,4,4′-TeBDE and 2,2′,4,4′,5-PeBDE. Degradation of DeBDE could be one explanation. The earlier production of lower brominated PBDE mixtures may also be a possibility. The observation of PBDEs and PBBs in sperm whales indicates that PBDEs and PBBs have reached deeper waters of the Atlantic Ocean and apparently are becoming global contaminants (de Boer *et al.*, 1998b). The PBDE concentrations found in North Sea dolphins (> 7 mg $kg^{-1}$ in blubber) are not as high as PCB concentrations found in North Sea dolphins (up to 128 mg $kg^{-1}$), but an increasing production may soon cause an increase in the environmental PBDE levels (de Boer *et al.*, 1998b; Boon *et al.*, 1997).

PBDEs have also been determined in human milk and tissue samples. A very interesting figure was produced by Norén and Meironyté (1998) (Figure A4.3). This figure shows the trend of PBDEs compared with those of the classic contaminants DDT and PCBs, showing that, whereas the DDTs and PCBs have decreased in human milk from Stockholm over the past 25 years, PBDEs are exponentially increasing. The concentration profiles of the PBDEs in these human milk samples were given by Meironyté *et al.* (1998), showing the highest concentrations for 2,2′,4,4′-TeBDE. DeBDE was not analysed. Darnerud *et al.* (1998) reported PBDE concentrations in human milk from Uppsala, Sweden. They also found 2,2′,4,4′-TeBDE in the highest concentrations (mean 2 $\mu g\ kg^{-1}$ lipid). Mean total PBDE values were around 4 $\mu g\ kg^{-1}$ lipid. De Boer *et al.* (1998a) found 2 $\mu g\ kg^{-1}$ (wet weight) 2,2′,4,4′-TeBDE and 4 $\mu g\ kg^{-1}$ 2,2′,4,4′,5-PeBDE in human adipose tissue of an Israeli who had been extensively exposed to vapours from a TV set. Lindström *et al.* (1998) determined 2,2′,4,4′-TeBDE in human adipose tissue from 77 individuals from Sweden (1995–1997). Mean concentrations were 4–16 $\mu g\ kg^{-1}$ lipid.

**Figure A4.3.a.** Trends in DDT concentrations in human milk from Stockholm, Sweden (Norén and Meironyté, 1998).

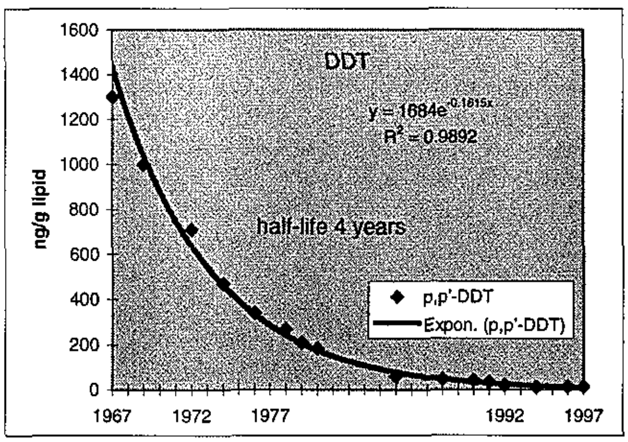


**Figure A4.3.b.** Trends in total dioxin 2,3,7,8-TCDD toxicity equivalents in human milk from Stockholm, Sweden (Norén and Meironyté, 1998).

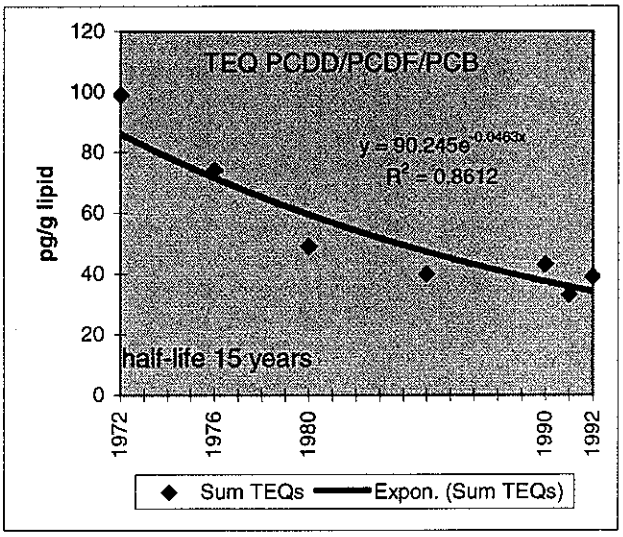
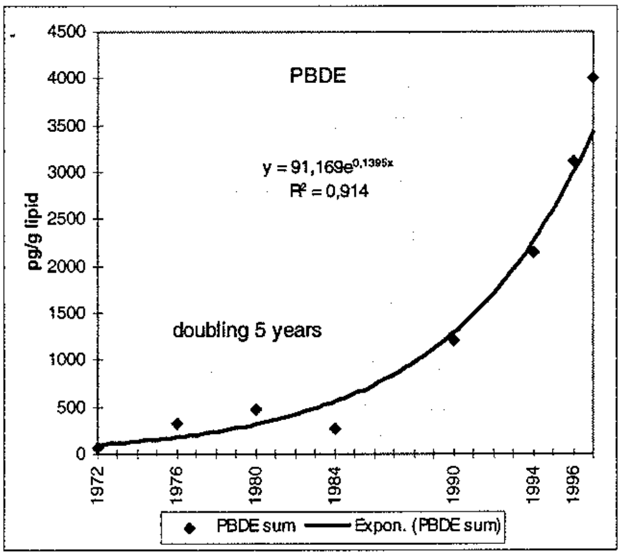



**Figure A4.3.c.** Trends in PBDE concentrations in human milk from Stockholm, Sweden (Norén and Meironyté, 1998).




## 10 TOXICOLOGY

### 10.1 Environmental Toxicity

#### 10.1.1 PBBs

Most data available concerning the effects of PBBs on organisms in the environment are data on farm animals from the Michigan disaster. The estimated average exposure of cows at a highly contaminated farm was 250 mg $kg^{-1}$ body weight (Fries, 1985). A few weeks after ingestion of contaminated cattle food, clinical signs were a reduction of about 50 % in food consumption (anorexia) and a decrease of about 40 % in milk production. Some cows showed an increased frequency of urination and lacrimation, and developed haematomas, abscesses, abnormal hoof growth, lameness, alopecia, hyperkeratosis, and cachexia. Several cows died within six months of exposure (Jackson and Halbert, 1974). The death rate in 6- to 18-month-old calves was much higher; 50 % died within six weeks (Jackson and Halbert, 1974; Robertson and Chynoweth, 1975). After the ingestion of a Fire Master mixture, hyperkeratosis and hair loss were seen in cattle, and lesions resembling chloro-acne were seen in rhesus monkeys after exposure to 50 mg $kg^{-1}$ in the diet (Safe, 1984). After 20 weeks, exposure to a dose of 2 mg per animal twice a week of Fire Master FF-1 also caused skin papillomas in previously initiated mice (Poland *et al.*, 1982). Fire Master BP-6 caused chronic and subchronic neuronal symptoms, such as irritation, changed behaviour, and a decreased muscular control (Safe, 1984).

In contrast to the observed toxicity in cattle, no clear health effects on the human population in Michigan could be correlated with PBB exposure. However, the follow-up period might not have been long enough for the development of cancer. In industry, it appears that chloro-acne-like lesions may develop in workers involved with the production of PBBs, and hypothyroidism can develop in workers exposed to BB-209 (WHO, 1994a). Further *in vitro* studies are necessary to identify toxic effects of PBBs. Controlled long-term feeding studies on cattle exposed to low doses of Fire Master did not reveal any adverse effects as indicated by food intake, clinical signs, clinico-pathological changes, or performance. Minks, guinea pigs, and monkeys appeared to be more susceptible to PBB toxicity (WHO, 1994a).

#### 10.1.2 PBDEs

There is almost no information available on the toxicity of PBDEs in organisms in the environment. A case study has been described of a young man who developed Yusho-like health effects after having watched a newly bought television set in a small non-ventilated insulated room for several hours a day during eight consecutive months at the age of 13. PBDEs were determined in a fat biopsy and a blood sample from this man at the age of 21. PBDE levels above the detection limit were only found in the fat biopsy. PBDEs, but also tetrabromobisphenol-A, were also found in different parts of the TV set (de Boer *et al.*, 1998a). Since eight years had elapsed between the possible exposure to high levels and the time of sampling, no definite answer could be given as to the role that PBDEs played in the adverse health effects on this man.

### 10.2 Acute Toxicity

#### 10.2.1 PBBs

Fire Master BP-6 appears to have a similar acute toxicity to rats as the PCB mixtures Aroclor 1254 and Kanechlor 500 (Pijnenburg *et al.*, 1995). The $LD_{50}$ values of commercial mixtures show a relatively low order of acute toxicity ($LD_{50}$ > 1 g $kg^{-1}$ body weight) in rats, rabbits, and quails, following oral or dermal administration. The toxicity of PBBs was higher with multiple-dose rather than single-dose administration. The few studies performed with commercial octabrominated biphenyl mixtures and BB-209 did not result in mortality in rats or fish. On the basis of the limited data available, octabrominated biphenyls and BB-209 appear to be less toxic than other lower brominated PBB mixtures, probably because they are less efficiently absorbed (WHO, 1994a).

#### 10.2.2 PBDEs

The acute toxicity of commercial PeBDE, DeBDE, and OcBDE for laboratory animals is low ($LD_{50}$ > 1 g $kg^{-1}$ body weight). BDE-209 and OcBDEs were also non-irritating to the skin. BDE-209 is neither irritating to the eyes of a rabbit nor did it have an effect on survival, body weight, or food consumption in feeding studies on rats and mice. No gross or microscopic pathological effects have been found (WHO, 1994b).

However, in short-term toxicity studies with octaBDEs, rats administered dietary levels of 100 mg $kg^{-1}$ had increased liver weights and showed microscopic changes in liver tissue. These liver tissue changes were even more severe at higher dosage levels, i.e., 1000 and 10 000 mg $kg^{-1}$ diet. OctaBDEs also gave minor eye irritation to rabbits (WHO, 1994b). In addition, pentaBDEs increased the liver/body weight ratio by 64 %, octaBDEs by 45 %, and BDE-209 by 25 % in a study in which a dose of 0.1 mM $kg^{-1}$ $day^{-1}$ was administered to male rats for 14 days (Carlson, 1980).

### 10.3 Mechanisms of Toxicity

The toxicity of PBB and PBDE congeners strongly depends on their molecular structure (WHO, 1994a, 1994b; Pijnenburg *et al.*, 1995).

Induction of the P4501A subfamily of cytochrome P450 is the precursor of a whole spectrum of possible effects at more integrated levels of biological structure: weight loss, thymus atrophy, and changes in the liver such as proliferation of the smooth endoplasmatic reticulum (location of the P450 system), increased RNA and protein content, decreased DNA content, cell necrosis, liver enlargement, and hepatic porphyria (Koster *et al.*, 1980; Render *et al.*, 1982; Jensen *et al.*, 1983).

#### 10.3.1 PBBs

The more toxic PBB congeners cause a decrease in thymus and/or body weight and produce pronounced histological changes in the liver and thymus. Categorization of halogenated biphenyls has been made on a structural basis. Category I comprises isomers and congeners lacking *ortho*-substituents (planar PBBs). Mono-*ortho*-substituted derivatives constitute the second category. Other PBBs (mainly those with two or more *ortho*-bromine substituents) have been organized into the third category. Congeners of category I tend to elicit the most severe effects, while the congeners of the second and third categories show decreasing toxicological changes. Within these categories, the degree of bromination may also influence toxicity. In all combinations tested, 3,3′,4,4′,5,5′-hexaBB (BB-169) was found to be the most toxic PBB (WHO, 1994a).

#### 10.3.2 PBDEs

PentaBDEs increased cytochrome P450 to a higher extent than did octaBDEs, while BDE-209 did not significantly increase cytochrome P450. 2,2′,4,4′-tetraBDE alone (6 and 18 mg $kg^{-1}$ body weight) induced EROD (CYP1A1) and MROD (CYP1A2) to a minor extent compared to PCBs in rats. A combination with chlorinated paraffins showed a synergistic effect on this induction. The induction of PROD (CYP2B) by 2,2′,4,4′-tetraBDE alone or in combination was as strong as that of the PCBs (Hallgren and Darnerud, 1998).

### 10.4 Genotoxicity

#### 10.4.1 PBBs

Several reports on the carcinogenicity of PBBs and PCBs have concluded that there are strong indications that these compounds are not themselves mutagenic, but that they promote the carcinogenicity of mutagenic compounds, such as nitrosamine and certain poly-aromatic hydrocarbons (PAHs) (Jensen *et al.*, 1983; Safe, 1984; Kavanagh *et al.*, 1985; Silberhorn *et al.*, 1990; Pijnenburg *et al.*, 1995). This is highly relevant, since in the marine environment halogenated compounds often co-occur with PAHs. The only lifetime study with a technical nonabromobiphenyl mixture was conducted on rats and mice. The lowest (oral) dose tested that continued to produce carcinogenic effects on rodents was 0.5 mg $kg^{-1}$ body weight per day, and the no observed effect level in a rat was < 0.15 mg $kg^{-1}$ body weight per day (Momma, 1986). The carcinogenicity of technical octabrominated biphenyl mixture and BB-209 has not been studied, although a number of chronic effects have been observed in experimental animals at doses of around 1 mg $kg^{-1}$ body weight per day during long-term exposure (WHO, 1994a).

### 10.4.2 PBDEs

In a carcinogenicity study in rats and mice, DeBDE was administered at dietary levels of up to 50 g $kg^{-1}$. An increased incidence of adenomas (but no carcinomas) was found in the livers of male rats receiving 25 g $kg^{-1}$ and female rats receiving 50 g $kg^{-1}$. In male mice, increased incidences of hepatocellular adenomas and/or carcinomas (combined) were found at 25 g $kg^{-1}$ and increased incidence of thyroid follicular cell adenomas/carcinomas (combined) at both dose levels. Female mice did not show any increase in tumour incidence. There was equivocal evidence for carcinogenicity in male and female rats and male mice only at dose levels of 25–50 g BDE-209 $kg^{-1}$ diet (NTP, 1986; Huff *et al.*, 1989). Since the results of all mutagenicity tests have been negative, it was concluded that BDE-209 is not a genotoxic carcinogen (WHO, 1994b). In 1990, the International Agency for Research on Cancer (IARC) concluded that there was limited evidence for carcinogenicity, indicating that BDE-209, at present exposure levels, does not present a carcinogenic risk for humans (WHO, 1994b).

The results for mutagenicity of pentaBDEs and BDE-209 were negative. Results of the mutagenicity tests of octaBDEs including an unscheduled DNA assay, *in vitro* microbial assays, and an assay for sister chromatid exchange with Chinese hamster ovary cells were also negative (WHO, 1994b). Very recently, the current levels of PBDEs in adipose tissue in humans from Sweden were correlated to the occurrence of non-Hodgkin's lymphoma (NHL), malignant melanoma, and other forms of cancer or *in situ* changes. All 77 patients had detectable levels of PBDEs in their adipose tissue, as represented by the concentrations of BDE-47. Only NHL patients showed significantly higher levels of BDE-47 (mean of 13.1 ng $g^{-1}$) than people without any malignancies (mean of 5.1 ng $g^{-1}$) (Lindström *et al.*, 1998). However, a mechanistic connection was not inferred.

## 10.5 Reproductive Effects

Both classes of compounds are listed as endocrine disruptors (Colburn *et al.*, 1993).

### 10.5.1 PBBs

Fire Master FF-1 caused a longer sexual cycle in monkeys (Pijnenburg *et al.*, 1995) and PBBs caused decreased egg production and nesting behaviour in Japanese quail (Aust *et al.*, 1987). One recent study reported that in mice PBB (diBBs and tetraBBs) reduced the *in vitro* fertilization rate at higher dosages. Furthermore, an increased incidence of abnormal two-cell embryos and degenerative oocytes was observed at 1–10 mg $ml^{-1}$ concentrations of PBBs (Kholkute *et al.*, 1994). PBBs also affect the regulation of steroid hormones. The extent depends on the species as well as the dose and duration of exposure.

### 10.5.2 PBDEs

DeBDE caused no teratogenic response in fetuses of rats intubated with 10–1000 mg $kg^{-1}$ $day^{-1}$ on gestation days 6–15. Fetal toxicity only occurred at 1000 mg $kg^{-1}$ as subcutaneous edema and a delayed ossification of normally developed bones of the fetal skull (Norris *et al.*, 1975). At high dose levels of octaBDEs (25 and 50 mg $kg^{-1}$ body weight) in rats, resorptions or delayed ossification of different bones and fetal malformations were observed (WHO, 1994b). In rabbits there was no evidence for teratogenic activity, but fetotoxicity was seen at a maternally toxic dose level of 15 mg octaBDE $kg^{-1}$ body weight (no observed effect level 2.5 mg $kg^{-1}$ body weight) (Breslin *et al.*, 1989). Test results for teratogenicity of PeBDE were negative (WHO, 1994b). Oral administration of a dietary dose of approximately 0.5 mg Bromkal 70-5 DE for 3.5 months to female sticklebacks, *Gasterosteus aculeatus*, resulted in a decreased spawning success (Holm *et al.*, 1993).

## 10.6 Effects on Thyroid Hormones

The interactions of both classes of brominated flame retardants with the thyroid hormone system seem to follow those of all major classes of polyhalogenated aromatic hydrocarbons (PHAHs) (Brouwer, 1998). Overall, three levels of interference in the thyroid system have been found for PHAHs, including the thyroid gland, thyroid hormone metabolism, and thyroid hormone transport. Effects on thyroid hormonal systems are important to follow, as these hormones play a crucial role in the development of many organs, e.g., the brain. Several enzyme systems involved in thyroid hormone metabolism have been found to be affected by PHAHs. A weak induction of cytochrome P4501A was measured after exposure of rats to 2,2′,4,4′-tetrabromodiphenyl-ether (Hallgren and Darnerud, 1998). The reduction of plasma levels of thyroxine (T4) seems to be caused especially by the hydroxylated metabolites of PHAHs through interaction with the transport protein transthyretin (TTR). Binding to TTR not only disrupts the T4 transport, with concomittant low plasma T4 levels, but may also result in the selective transport of the hydroxylated PHAHs across the blood-brain barrier and the placental barrier. TTR has been suggested to play a major role in mediating the delivery of T4 from the mother to the fetus across the placental barrier, but also across the blood-brain barrier, where T4 is locally converted to T3, which is absolutely essential for, e.g., brain development. Competitive inhibition of thyroid binding to choroid plexus by several hydroxylated PCBs and a hydroxylated PCB metabolite was shown to occur in rats (Sinjari *et al.*, 1998). The possible impact of the high accumulation of phenolic PHAHs on the development of fetal brain, behaviour, and reproductive organs and functions is the focus of ongoing research. Planarity of the hydroxy-PHAH molecules is not a requirement for binding to TTR. Many effects on the thyroid hormone system are primarily caused by the hydroxy metabolites of PHAHs. In this respect, it is noteworthy that, of a number of PBB and PBDE congeners investigated, only 4,4′-dibromobiphenyl (BB-15) was metabolized in *in vitro* assays with microsomal preparations from seals, cetaceans, and Laysan albatross (Watanabe *et al.*, 1994; de Boer *et al.*, 1998b). In contrast, BDE-47 and BDE-99 were rapidly metabolized in an experiment with rats and mice (Klasson-Wehler *et al.*, 1996). Thus, large species differences in sensitivity towards this mechanism of toxicity may exist.

#### 10.6.1 PBBs

Rats and pigs showed dose-related decreases in serum thyroxine and triiodothyronine. There was also a pronounced influence of PBBs on vitamin A storage above a no observed effect level of 0.1 mg $kg^{-1}$ body weight per day (WHO, 1994a). This effect probably also involves hydroxy metabolites, since this is the case with PCBs.

#### 10.6.2 PBDEs

Hyperplasia of the thyroid has been observed (Great Lakes Chemical Corporation, 1987). Similar observations have been reported for pentaBDEs (WHO, 1994b) and BDE-209 (Norris *et al.*, 1975). Exposure of rats for 14 days to a daily dose of 18 mg 2,2′,4,4′-tetraBDE $kg^{-1}$ body weight resulted in lowered levels of free thyroxine in blood plasma. Lower doses showed no such effect (Hallgren and Darnerud, 1998).

### 10.7 Immunosuppression

#### 10.7.1 PBBs

PBBs cause immunosuppression at levels that also cause a number of the other toxic effects described (Pijnenburg *et al.*, 1995).

#### 10.7.2 PBDEs

The immunotoxic potential of the PBDE mixture Bromkal 70-5 DE was compared to the technical PCB mixture Aroclor 1254 and the potential of the single congener 2,2′,4,4′-tetraBDE to that of 2,3,3′,4,4′-pentachlorobiphenyl (CB105) in rats and mice. In mice, all compounds were immunotoxic, but in rats only the PCBs were. This points to the existence of a considerable range of species differences (Darnerud and Thuvander, 1998).

### 10.8 Hepatic Porphyria

PBBs produced porphyria in rats and male mice at doses as low as 0.3 mg $kg^{-1}$ body weight per day.

## 11 CONCLUSIONS

In summary, it can be concluded that the acute toxicity of brominated flame retardants is relatively low. The long-term effects on the balance of endocrine systems seem to present the greatest danger of these compounds. These endocrine effects need further consideration, since the majority of animals and man are exposed to these brominated flame retardants. The exposure range for humans via food was calculated as 0.2–0.7 µg per day (Darnerud *et al.*, 1998).

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ANNEX 5

# ICES DATA FOR AN HOLISTIC ANALYSIS OF FISH DISEASE PREVALENCE

## 1 INTRODUCTION

At its 1998 meeting, the ICES Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) reviewed the results of a statistical analysis of the ICES fish disease data; a written report was presented providing information on location-specific temporal trends in the prevalence of lymphocystis, epidermal hyperplasia/papilloma, and skin ulcers in dab (*Limanda limanda*) and lymphocystis and skin ulcers in flounder (*Platichthys flesus*), as well as a comparison of trends found at different geographical locations (ICES, 1999).

The results of the analysis revealed marked spatial differences with respect to both the absolute levels and the temporal changes of the disease prevalences. Furthermore, areas characterized either by decreasing, increasing or stable temporal trends over the past years were identified. In some areas, temporal changes in the prevalence of two or more diseases were similar, thereby indicating the presence of common underlying ecological factors affecting the disease prevalence. However, the results of the analysis did not provide any information on possible causes of the observed trends, since potential explanatory factors known or suspected to be involved in disease aetiology and pathogenesis were not included in the analysis at that time.

WGPDMO emphasized that the integration of the ICES fish disease data and other types of data (e.g., contaminant, oceanographic, and fisheries-related data) would constitute a subsequent next step in the attempt to analyse the ICES fish disease data in a more holistic way, with the aim of obtaining better insight into cause-effect relationships between diseases and environmental factors. Since ICES has established different data banks with relevant data, it was recommended that available data should be assessed with respect to their usefulness for such a holistic approach.

When reviewing the 1998 WGPDMO report, the ACME endorsed the view that a more holistic statistical analysis is desirable and that the various ICES data banks could serve as a suitable data pool from which the information required could be extracted. The ACME emphasized that the first step taken, before any further actions are decided upon, should be to obtain a detailed overview of the data availability and compatibility. This should be done in close collaboration between selected WGPDMO members and the ICES Secretariat. A second step, a pilot study, could be subsequently started, using a selected subset of suitable data extracted from the ICES databases, in order to assess the practicality and perspectives of a future holistic data analysis.

The results of intersessional activities carried out according to the above recommendations, prior to the 1999 WGPDMO meeting, are presented below. The first part of the report describes the outcome of the assessment of the ICES data banks with respect to the types and amounts of data available, with particular emphasis on spatial and temporal data coverage and overlap. In the second part, preliminary results of a multivariate statistical analysis of a selected subset of data are presented. The third part provides conclusions and perspectives, e.g., by focusing on data limitations identified and on ways to accomplish a more comprehensive analysis.

## 2 OVERVIEW OF AVAILABLE DATA

Information on the ICES data banks from which data can be extracted for a holistic analysis and on strategies applied to obtain an overview of available data is described below. In addition, some of the data are presented in order to demonstrate temporal trends.

### 2.1 ICES Data Banks

ICES data considered relevant for a holistic analysis are available from the following data banks:

1) ICES Environmental Data Centre;

2) ICES Oceanographic Data Centre;

3) ICES Fishery Data Banks.

ICES provides a detailed overview (partly interactive) of the data included on its website http://www.ICES.dk. Most of the information on the data banks detailed below was extracted from this website.

### A ICES Environmental Data Centre

The ICES Environmental Data Centre contains data on:

- contaminants in marine invertebrates, fish, birds, and mammals (approximately 275 000 records);
- contaminants in sea water (approximately 280 000 records);
- contaminants in sediments (approximately 80 000 records);
- biological effects of contaminants: EROD, oyster embryo bioassay (approximately 4000 records);
- fish disease prevalences (approximately 110 000 records);

- supporting data: nutrients, oxygen, temperature, salinity;
- quality assurance (QA) information.

### B ICES Oceanographic Data Centre

The ICES Oceanographic Data Centre maintains two data banks in the ICES Secretariat:

- the ROSCOP data bank (information on cruise activities);
- the hydrochemical data bank (temperature, salinity, nutrients, oxygen, etc.).

In addition to these holdings, there is access to a number of project data sets (including oceanographic data from the International Young Fish/Bottom Trawl Surveys).

### C ICES Fishery Data Banks

The ICES Fishery Data Banks include five fisheries-related data banks:

- STATLANT 27A (official statistics on nominal catches of fish and shellfish);
- ICES Fisheries Assessment Package (for use by approximately twenty working groups for ICES stock assessments—includes catches in tonnes, fishing effort, catch in number at age, and relevant biological data);
- International Bottom Trawl Survey (IBTS) (results from an international survey conducted each year in the North Sea since 1970 which provides an annual index of abundance by ICES Statistical Rectangle; additional data on temperature, salinity, nutrients);
- North Sea data bank (contains details of catches and fishing effort originally set up by the EC);
- North Sea multispecies data bank (contains stomach content data for each of the main predatory fish species in the North Sea—for use in multispecies models).

## 2.2 Strategies Applied to Obtain an Overview of ICES Data

Since the ICES data that are available and potentially relevant for a holistic analysis are overwhelming in terms of parameters measured and results of measurements, it was considered impossible to present a full overview on their spatial and temporal distribution patterns. It was, therefore, decided to extract some of the data by using *a priori* selection criteria that were mainly based on the availability of ICES disease prevalence data for common dab (*Limanda limanda*).

### 2.1.1 Selection criteria

***Sites***

Three North Sea areas (extended German Bight, Dogger Bank, Firth of Forth) were selected for which a considerable amount of disease data are available and which differ both in the absolute prevalence levels and the temporal trends recorded over the past years. As shown in Figure A5.1, the areas were relatively large and consisted of four to nine ICES Statistical Rectangles in order to obtain sufficient data for the subsequent statistical analysis.

***Time span***

Since the fish disease data date back to 1981, this was the year used as the starting point for the temporal overview.

***Parameters***

From the ICES Fishery Data Banks, data on catch per unit effort (CPUE) for dab (all specimens, independently of size) derived from the International Young Fish/Bottom Trawl Survey (IYFS/IBTS) were selected. Originally, it was also planned to incorporate fishing effort data (STECF data, EU Scientific Technical and Economic Committee for Fisheries). However, since they are not structured in an easily accessible way (e.g., effort data are only available for a short time period and separately for sixty different fishing fleets), they were excluded. From the ICES Oceanographic Data Centre, information on water temperature, salinity, dissolved oxygen, total phosphorus, phosphate, ammonium, nitrite, nitrate, silicate, and chlorophyll were considered, partly derived from the International Young Fish/Bottom Trawl Survey. From the ICES Environmental Data Centre, data on contaminants (Pb, Hg, Cd, HCH, HCB, CB118, CB153, *o,p* DDT) in water, sediments, dab muscle and liver, and blue mussel (*Mytilus edulis*) were selected.

After the selections were made, the ICES Secretariat was contacted and the data were requested in electronic form. According to the ICES data policy, raw data are not available as this would require permission from the data originators. Instead, aggregated data, consisting of, e.g., calculated mean values, were provided by the Secretariat.

## 2.3 Brief Overview

Figures A5.2–A5.4 provide an overview of the data available according to the selection criteria described above. In addition to the parameters mentioned, ICES disease prevalence data for female dab, size group 20–24 cm, are also included. The figures clearly show that most data are available for Area 1 (extended German Bight), followed by Area 2 (extended Dogger Bank) and, finally, Area 3 (extended Firth of Forth). Apart from the disease data, there is relatively good temporal coverage of data on CPUE, water temperature, salinity, and nutrients for all three areas for almost the whole period, but only few data are available on contaminants in water (with the exception of Area 1, where data are available from 1985 onwards), sediments (with the exception of data from 1990–1992), and dab (with the exception of Areas 1 and 2 from 1990–1996).

In order to provide an overview of temporal trends, Figures A5.5a–A5.5f show the data for selected parameters measured in Area 1, the extended German Bight. Both the observed values and the values derived from interpolation in preparation of the subsequent analysis (see below) are shown, with their corresponding confidence intervals. For contaminants in dab, only data for female dab, size group ≥ 20 cm, are included since they correspond to the standard disease prevalence data used in previous statistical analyses (females, 20–24 cm; see ICES, 1999). In the ICES Environmental Data Centre, contaminant and disease data for other size groups and for males are also available.

# 3 CASE STUDY (STATISTICAL ANALYSIS)

As requested by ACME, a case study was performed for which only the data set from Area 1 (extended German Bight) was used since it constituted the largest set with the best temporal coverage of parameters.

## 3.1 Material and Methods

Figures A5.2–A5.4 reveal that not all data to be compared were recorded at the same time and, therefore, some values had to be interpolated prior to analysis. Since the intention was to relate the data to the disease prevalence data, interpolations were calculated for those time points (days) for which disease data were available. A kernel smoother with Gaussian kernel was used to interpolate and to remove random fluctuation. The smoother bandwidth was selected by generalized cross-validation. No extrapolation outside the time range covered by real observations was done. Pointwise confidence intervals (upper and lower lines in Figures A5.5a–A5.5f) for interpolated values were derived as two standard deviations around smoothed values.

As a first approach to identify relationships between the disease prevalence and the parameters listed in Section 2.2, univariate logistic models involving observed and interpolated values were fitted (see Table A5.1).

As a second approach, a multivariate logistic model was fitted for each of three scenarios comprising different sets of parameters (see Table A5.2). In the long-term model, parameters with observations for nearly the entire range from 1981–1997 were included. The medium and short-term models contained additional parameters that covered approximately one half and one third, respectively, of this range. A stepwise selection was used to identify parameters with the highest explanatory values. A multivariate analysis comprising all parameters simultaneously was considered inappropriate at this stage as it would have required either massive extrapolations or would have been restricted to a very narrow time span.

## 3.2 Results and Discussion

The results of the univariate analysis are given in Table A5.1. A comparison of the ratio of the number of observations ($n_o$) and the number of interpolated data points ($n_i$) used in the analysis for each parameter provides not only an overview of the data availability but also a rough indicator of the reliability of the results of the analysis; if the balance is towards the number of observations, interpolated data can be considered more valid than if the balance is towards the interpolated values.

For each parameter and disease tested, the table includes information on the direction of the relationship and the significance levels. In total, a significant relationship was identified in 31 of 66 cases, which could be taken as an indicator of the multifactorial aetiology of the diseases considered. Water temperature was the parameter with the strongest impact and the prevalences of all three diseases were positively and highly significant when related to temperature. CPUE was also significantly correlated to the prevalence of the diseases. However, a positive relationship was only found for lymphocystis and epidermal hyperplasia/papilloma, while a negative relationship was found for acute/healing skin ulcerations. Information on the significance of other parameters can be taken directly from the table.

Table A5.2 provides information on the parameters found significantly correlated to disease prevalence by using a stepwise multivariate analysis. Assuming an 'ideal', e.g., consistent, relationship between disease prevalence and certain parameters, one would have expected that terms with a highly significant impact in the long-term model would also appear in the medium-term and short-term models for the respective diseases. However, this was only the case for water nitrate and epidermal hyperplasia/papilloma. Other significant terms occurring at least twice were salinity and cadmium in *M. edulis* (lymphocystis), CB153 in *M. edulis* and cadmium in unfiltered sea water (epidermal hyperplasia/-papilloma), and cadmium in unfiltered sea water and CPUE (acute/healing skin ulcerations).

The inconsistency between parts of the univariate and the multivariate analyses with regard to the parameters identified as significant (e.g., for water temperature) is at least partially due to the fact that these analyses referred to different time ranges. Furthermore, some of the parameters were found to be highly correlated (e.g., water temperature with CPUE, nitrate, and phosphate), which means that, in the multivariate models, certain parameters might replace others in order to explain variation in prevalence.

For interpretation of the results of the analyses, it has to be taken into account that a considerable part of the data used was derived from data interpolation, which is always problematic since one does not know how the parameters have behaved in between two sampling dates. With increasing distance between sampling dates, this problem becomes more serious and it is self-evident that the reliability of results from any statistical analysis suffers from this. Furthermore, automatic interpolation without adaptation to the type of data used might lead to biased data, e.g., if data series characterized by pronounced seasonal variation are not consistent in terms of temporal coverage of data points. An example from the existing ICES data set is the measurement of water temperature (Figure A5.5a). For the period 1981–1991, only winter data are available and interpolated values used for the analysis were generally low. From 1991, however, spring/summer values are also available and, therefore, interpolated values were considerably higher although winter values remained at about the same level as in previous years. This example clearly demonstrates that, for further analyses, it will be more appropriate to employ specific parameter-oriented interpolation techniques.

## 4 CONCLUSIONS AND PERSPECTIVES

First of all, with regard to the activities of the WGPDMO during previous years, the data overview reveals that the fish disease data constitute one of the most comprehensive and consistent data sets in the ICES Environmental Data Centre, in terms of both spatial and temporal coverage.

In contrast, there is a striking lack of other data, in particular for contaminants in biota and sediments, creating major problems in the initiated holistic data analysis. However, there are undoubtedly more data around held by national data banks. Therefore, ICES Member Countries should be encouraged to submit these data to the ICES Environmental Data Centre by using standard procedures already developed by ICES and, therefore, applicable without major effort.

The availability of additional data would improve the spatial and temporal data coverage and would, therefore, possibly facilitate an analysis based on smaller geographical areas than those used in the present case study. Large areas create problems since conditions normally are not the same over the entire area. For instance, Area 1 (extended German Bight) includes coastal, estuarine, and offshore areas and a comparison of, e.g., contaminant levels in mussels from estuarine areas with disease prevalences of dab collected offshore can only be regarded as a rough approach to identify possible relationships. Furthermore, additional data could minimize the need for temporal interpolation and related problems (see above).

Despite the shortcomings identified, the results of the analysis seem to be promising since, for a number of parameters included, a close relationship with variation in disease prevalence could be identified. It is, therefore, concluded that further activities with respect to enhancing the data basis and improving the models applied and the methods for statistical analysis are desirable and should be initiated.

## 5 ACKNOWLEDGEMENT

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**Abbreviations used in the figures and tables, if not further specified**

| | |
|---|---|
| CPUE | catch per unit effort (number of fish per hour of trawling) |
| DIS OXY | dissolved oxygen |
| F1 | females, size group 10–14 cm |
| F2 | females, size group 15–19 cm |
| F3 | females, size group 20–24 cm |
| F4 | females, size group ≥ 25 cm |
| F20 | fraction < 20 µm |
| F63 | fraction < 63 µm |
| LI | liver tissue |
| LIMA | *Limanda limanda* |
| M1 | males, size group 10–14 cm |
| M2 | males, size group 15–19 cm |
| M3 | males, size group 20–24 cm |
| M4 | males, size group ≥ 25 cm |
| MU | muscle tissue |
| MYTI | *Mytilus edulis* |
| P TOTAL | total phosphorus |
| SB | soft body tissue |
| SED | sediment |
| U00 | unfractionated |
| WAT | water |
| WAT A | filtered sea water |
| WAT B | unfiltered sea water |

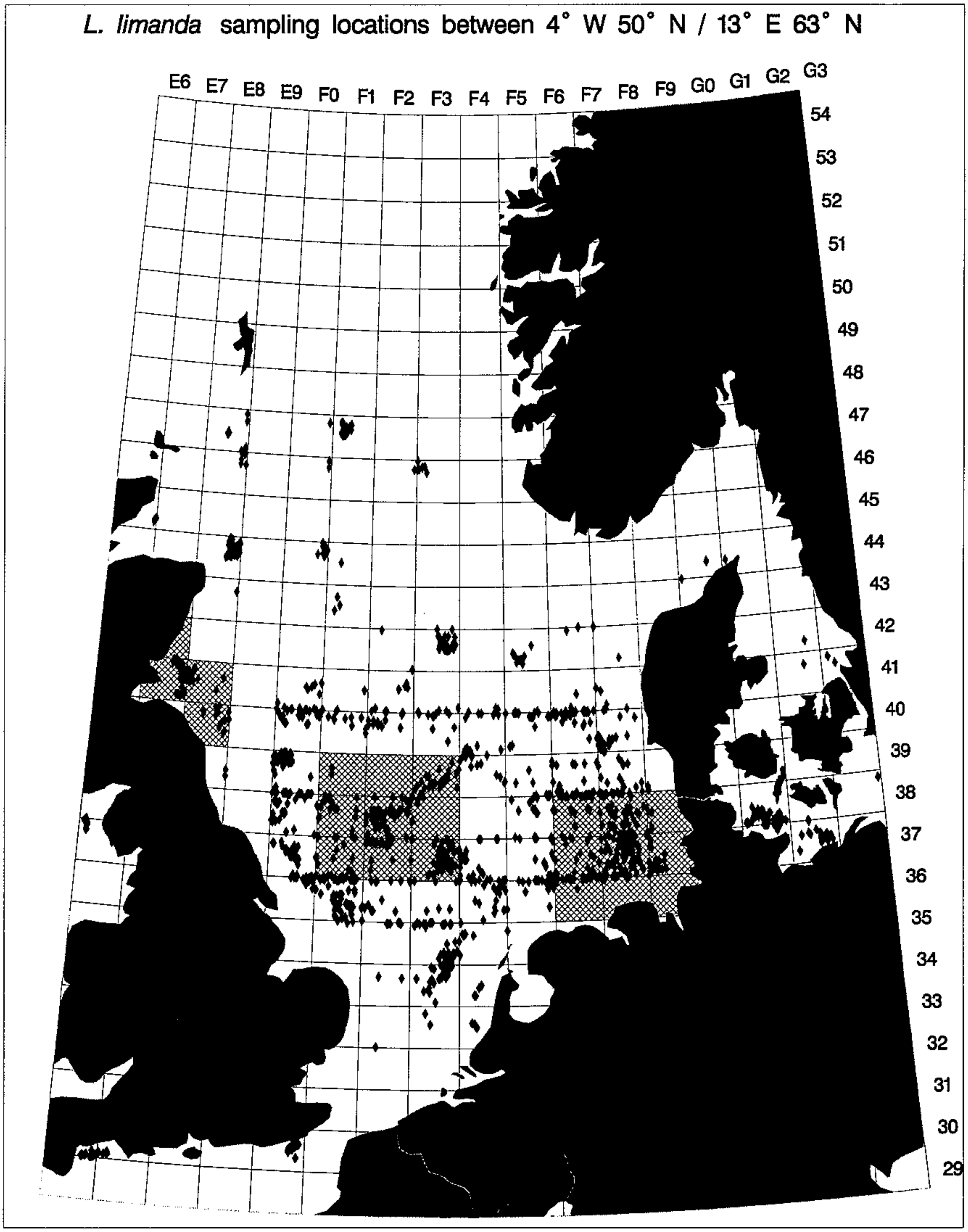
**Table A5.1.** Area 1 (extended German Bight): case study on the relationship between the prevalence of dab (*L. limanda*) diseases and parameters available from the ICES data banks, results of a univariate analysis (if $p < 0.05$, a significant relationship existed). $n_o$: number of observations; $n_i$: number of interpolations; dir: direction of relationship; p: significance level.

| Parameter | $n_o$ | $n_i$ | Lymphocystis | | Epidermal hyperplasia/ papilloma | | Acute/healing ulcerations | |
|---|---|---|---|---|---|---|---|---|
| | | | dir | p | dir | p | dir | p |
| *W TEMPERATURE* | 261 | 98 | + | < 0.0001 | + | < 0.0001 | + | < 0.0001 |
| *W SALINITY* | 239 | 98 | + | < 0.0001 | + | < 0.0001 | - | 0.8911 |
| *W PHOSPHATE* | 48 | 71 | + | 0.1688 | - | 0.0638 | - | 0.0418 |
| *W DIS OXY* | 20 | 29 | + | 0.0013 | + | 0.1358 | - | 0.0733 |
| *W NITRATE* | 48 | 71 | + | 0.0362 | - | 0.7437 | - | 0.0832 |
| | | | | | | | | |
| *WAT B CB153* | 120 | 43 | + | 0.2149 | - | 0.7491 | + | 0.9132 |
| *WAT B HCB* | 114 | 43 | - | 0.6210 | - | 0.7535 | - | < 0.0001 |
| *WAT B Cd* | 185 | 52 | - | 0.5397 | + | 0.2417 | - | < 0.0001 |
| *WAT B Hg* | 186 | 52 | - | 0.0066 | - | 0.0051 | - | 0.0555 |
| | | | | | | | | |
| *SED F63 CB153* | 5 | 42 | + | 0.5098 | - | 0.0098 | + | 0.2030 |
| *SED F63 HCB* | 3 | 27 | - | 0.0006 | - | 0.0534 | + | 0.0317 |
| *SED F63 Cd* | 7 | 34 | + | 0.2290 | + | 0.7014 | + | 0.9162 |
| *SED F63 Hg* | 8 | 42 | + | 0.0114 | - | 0.4048 | + | 0.8843 |
| | | | | | | | | |
| *MYTI SB CB153* | 36 | 96 | + | < 0.0001 | + | < 0.0001 | + | 0.4567 |
| *MYTI SB HCB* | 33 | 39 | - | 0.0183 | - | 0.0544 | - | 0.0128 |
| *MYTI SB Cd* | 50 | 96 | + | 0.1833 | + | 0.1119 | - | 0.0238 |
| *MYTI SB Hg* | 51 | 96 | + | < 0.0001 | + | < 0.0001 | - | 0.1370 |
| | | | | | | | | |
| *LIMA LI CB153* | 4 | 13 | + | 0.0192 | - | 0.4267 | + | 0.9582 |
| *LIMA LI HCB* | 3 | 5 | + | 0.1218 | - | 0.7319 | + | 0.6986 |
| *LIMA LI Cd* | 5 | 28 | + | 0.0235 | + | 0.8689 | - | 0.0115 |
| *LIMA MU Hg* | 5 | 28 | + | 0.0019 | - | 0.8375 | - | 0.0064 |
| | | | | | | | | |
| *CPUE* | 295 | 97 | + | < 0.0001 | + | < 0.0001 | - | 0.0060 |

**Table A5.2.** Area 1 (extended German Bight): case study on the relationship between the prevalence of dab (*L. limanda*) diseases and parameters available from the ICES data banks, results of a multivariate analysis. $n_i$= number of interpolations (data values) remaining for the analysis; dir = direction of relationship; p = significance level.

| **Long-term model** | **Parameters included in the model** | | |
|---|---|---|---|
| | W TEMPERATURE, W SALINITY, W PHOSPHATE , W NITRATE,<br>WAT B CB153, WAT B HCB, WAT B Cd, WAT B Hg,<br>MYTI SB CB153, MYTI SB Cd, MYTI SB Hg,<br>CPUE | | |
| $n_i$ | 43 | | |
| **Disease** | **Significant terms** | **dir** | **p** |
| Lymphocystis | W SALINITY<br>MYTI SB Cd | -<br>- | < 0.0001<br>< 0.0001 |
| Epidermal hyperplasia/ papilloma | W NITRATE<br>WAT B Cd<br>MYTI SB CB153<br>MYTI SB Cd | +<br>+<br>-<br>- | 0.0003<br>< 0.0001<br>< 0.0001<br>< 0.0001 |
| Acute/healing ulcerations | WAT B Cd<br>CPUE | -<br>- | 0.0008<br>0.0163 |
| **Medium-term model** | **Parameters included in the model** | | |
| | W TEMPERATURE, W SALINITY, W PHOSPHATE , W NITRATE,<br>WAT B CB153, WAT B HCB, WAT B Cd, WAT B Hg,<br>SED F63 CB153, SED F63 Hg,<br>MYTI SB CB153, MYTI SB Cd, MYTI SB Hg,<br>CPUE | | |
| $n_i$ | 33 | | |
| **Disease** | **Significant terms** | **dir** | **p** |
| Lymphocystis | W SALINITY<br>MYTI SB Cd<br>CPUE | -<br>-<br>- | < 0.0001<br>< 0.0001<br>0.0213 |
| Epidermal hyperplasia/ papilloma | W NITRATE<br>WAT B CB153<br>SED F63 CB153<br>MYTI SB CB153 | +<br>-<br>-<br>+ | < 0.0001<br>0.0089<br>< 0.0001<br>0.0004 |
| Acute/healing ulcerations | CPUE | - | 0.0016 |
| **Short-term model** | **Parameters included in the model** | | |
| | W TEMPERATURE, W SALINITY, W PHOSPHATE , W NITRATE,<br>WAT B CB153, WAT B HCB, WAT B Cd, WAT B Hg,<br>SED F63 CB153, SED F63 Hg,<br>MYTI SB CB153, MYTI SB HCB, MYTI SB Cd, MYTI SB Hg,<br>LIMA LI Cd, LIMA MU Hg,<br>CPUE | | |
| $n_i$ | 23 | | |
| **Disease** | **Significant terms** | **dir** | **p** |
| Lymphocystis | W NITRATE<br>MYTI SB CB153<br>LIMA LI Cd | +<br>+<br>- | 0.0093<br>< 0.0001<br>0.0070 |
| Epidermal hyperplasia/ papilloma | W NITRATE<br>WAT B Cd | +<br>+ | 0.0005<br>0.0288 |
| Acute/healing ulcerations | WAT B Cd | - | 0.0070 |

**Figure A5.1.** Location of the three North Sea areas for which an overview of available ICES data is presented (single marks indicate positions for which prevalence data of dab (*L. limanda*) diseases are available).



**Figre A5.2.** Area 1 (extended German Bight): overview of ICES data available for a holistic analysis of fish disease data (extract).

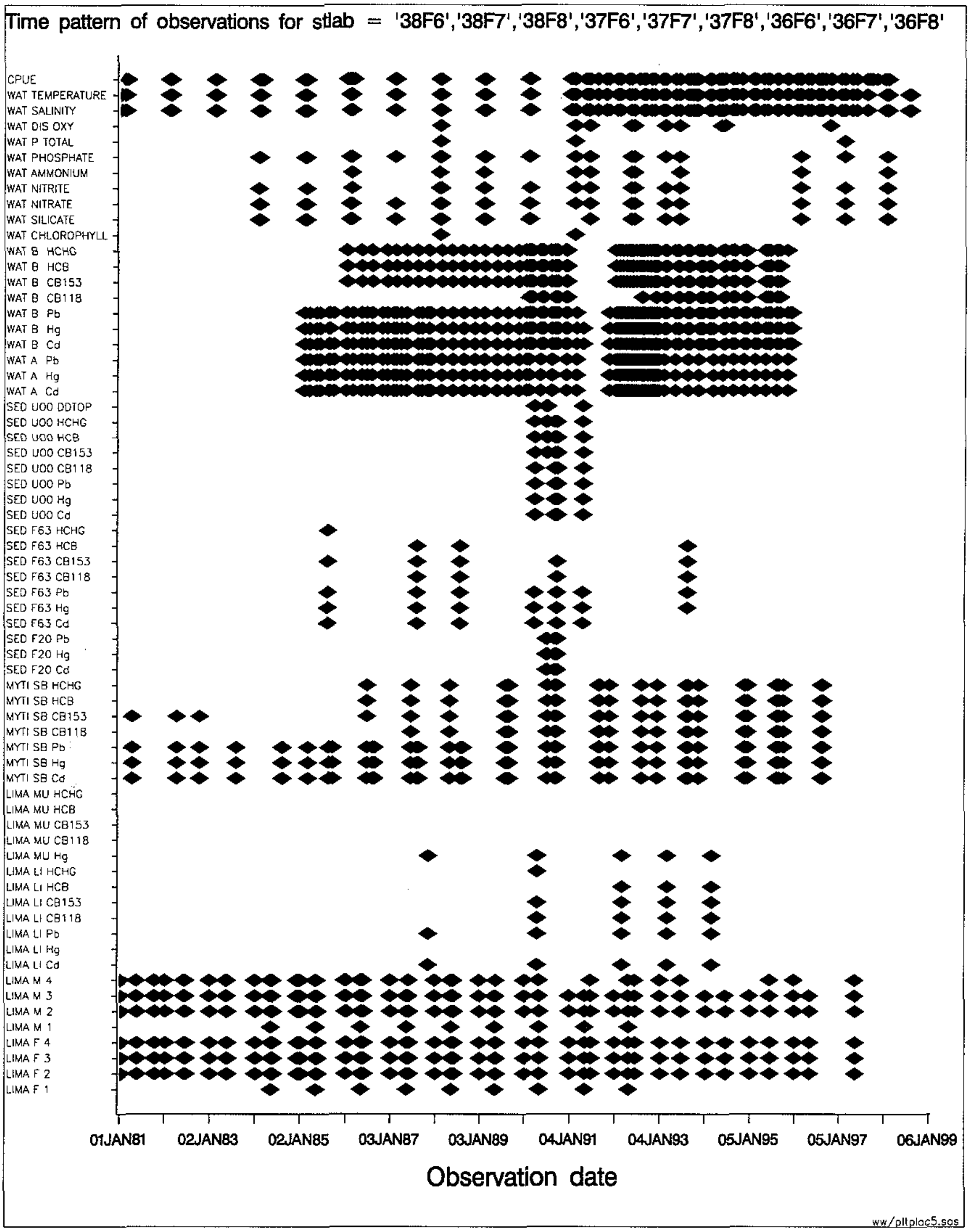


**Figure A5.3.** Area 2 (extended Dogger Bank): overview of ICES data available for a holistic analysis of fish disease data (extract).

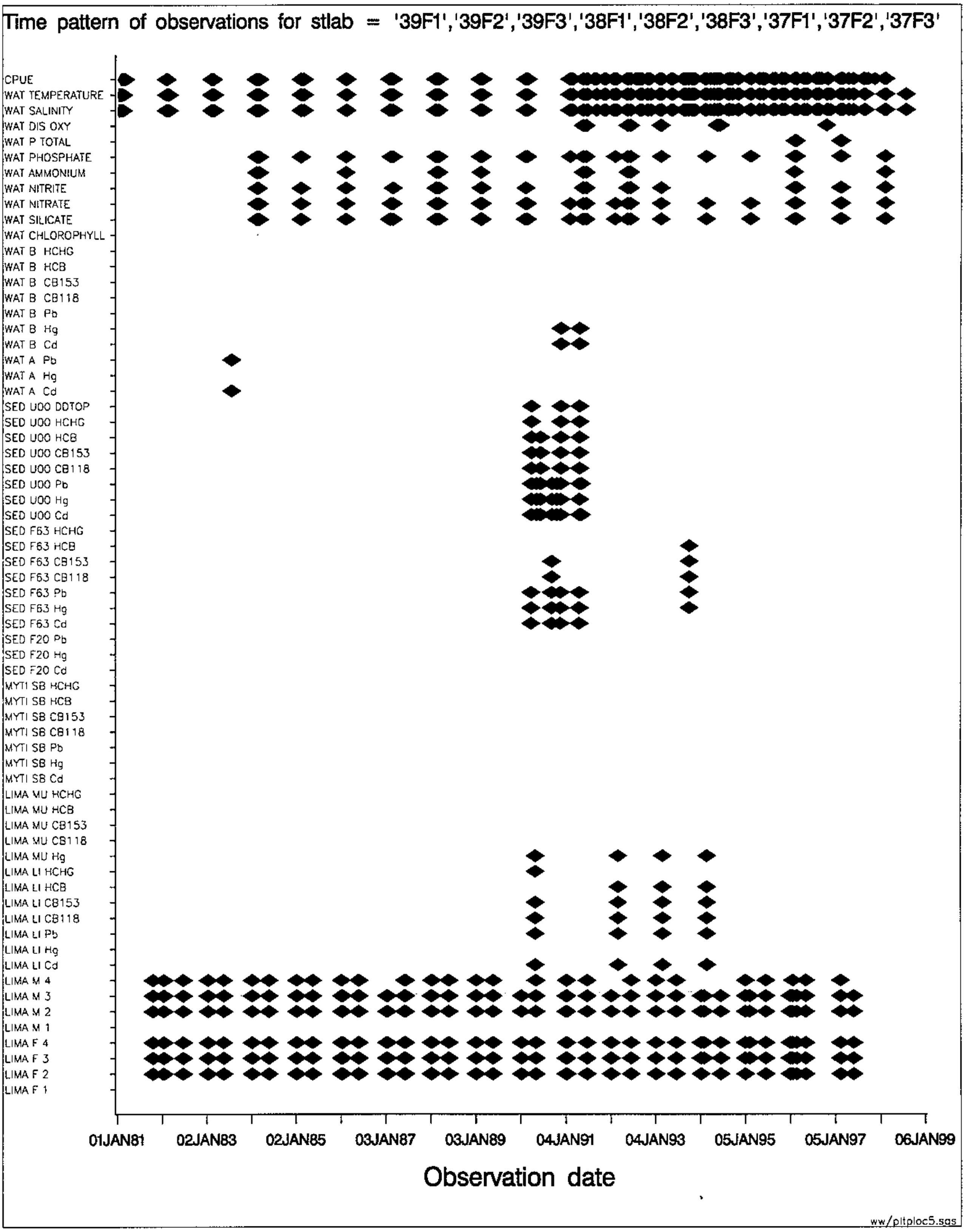


**Figure A5.4.** Area 3 (extended Firth of Forth): overview of ICES data available for a holistic analysis of fish disease data (extract).

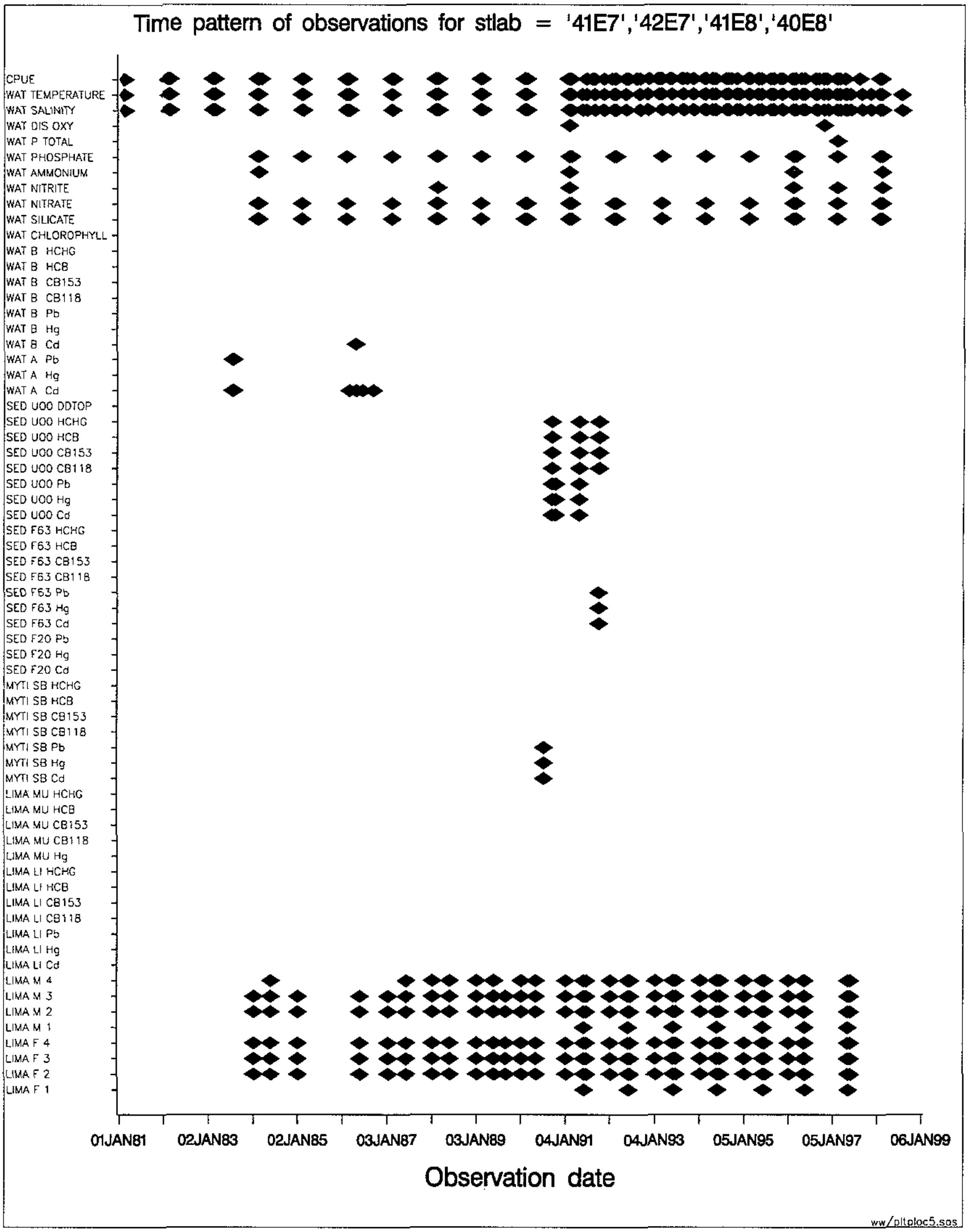


**Figure A5.5.a.** Area 1 (German Bight, extended): water data from the ICES Oceanographic Data Centre (extract) (filled circles: empirical data, empty circles: interpolated values).

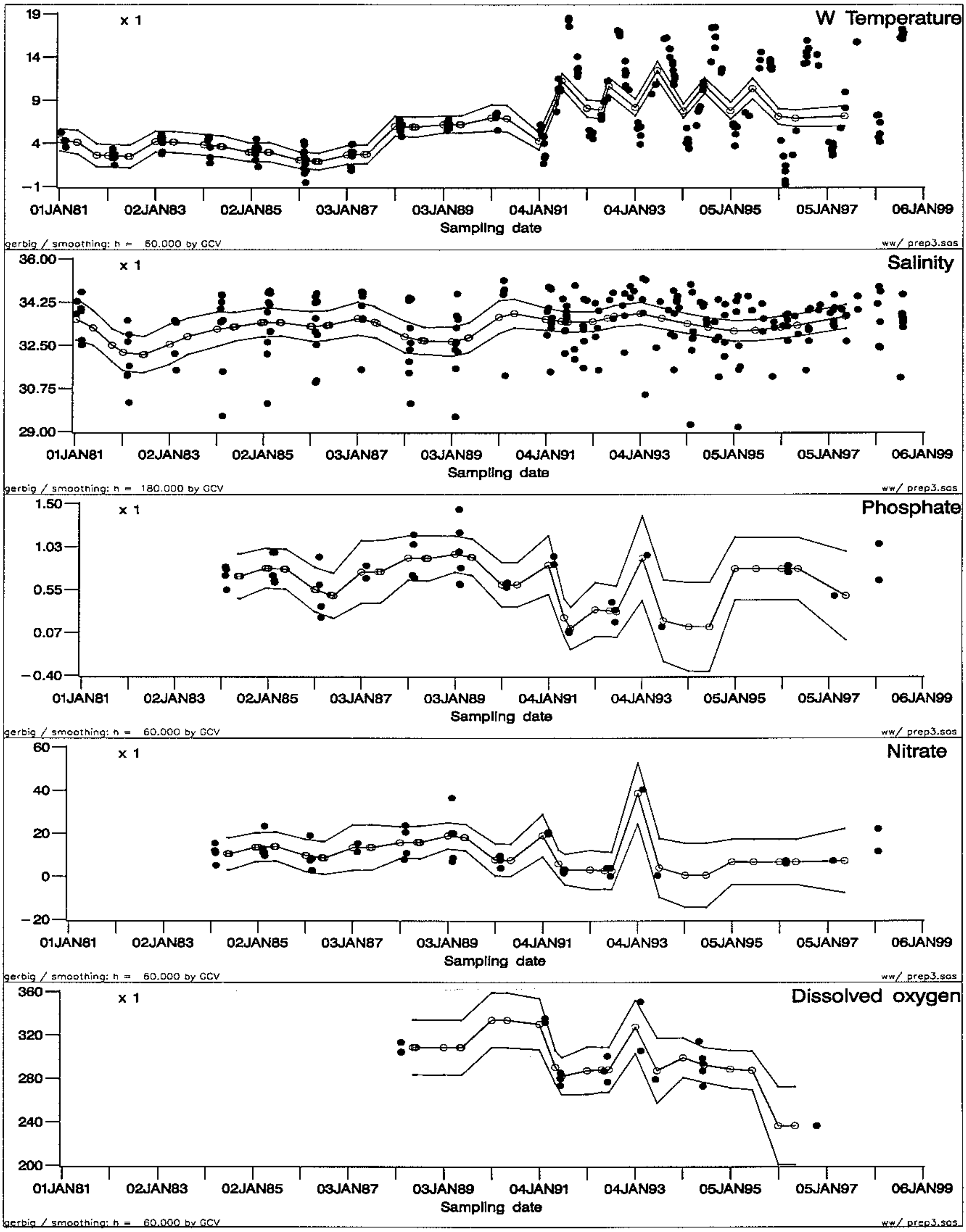


**Figure A5.5.b.** Area 1 (extended German Bight): data on contaminants in sea water (unfiltered) from the ICES Environmental Data Centre (extract) (filled circles: empirical data; empty circles: interpolated values).

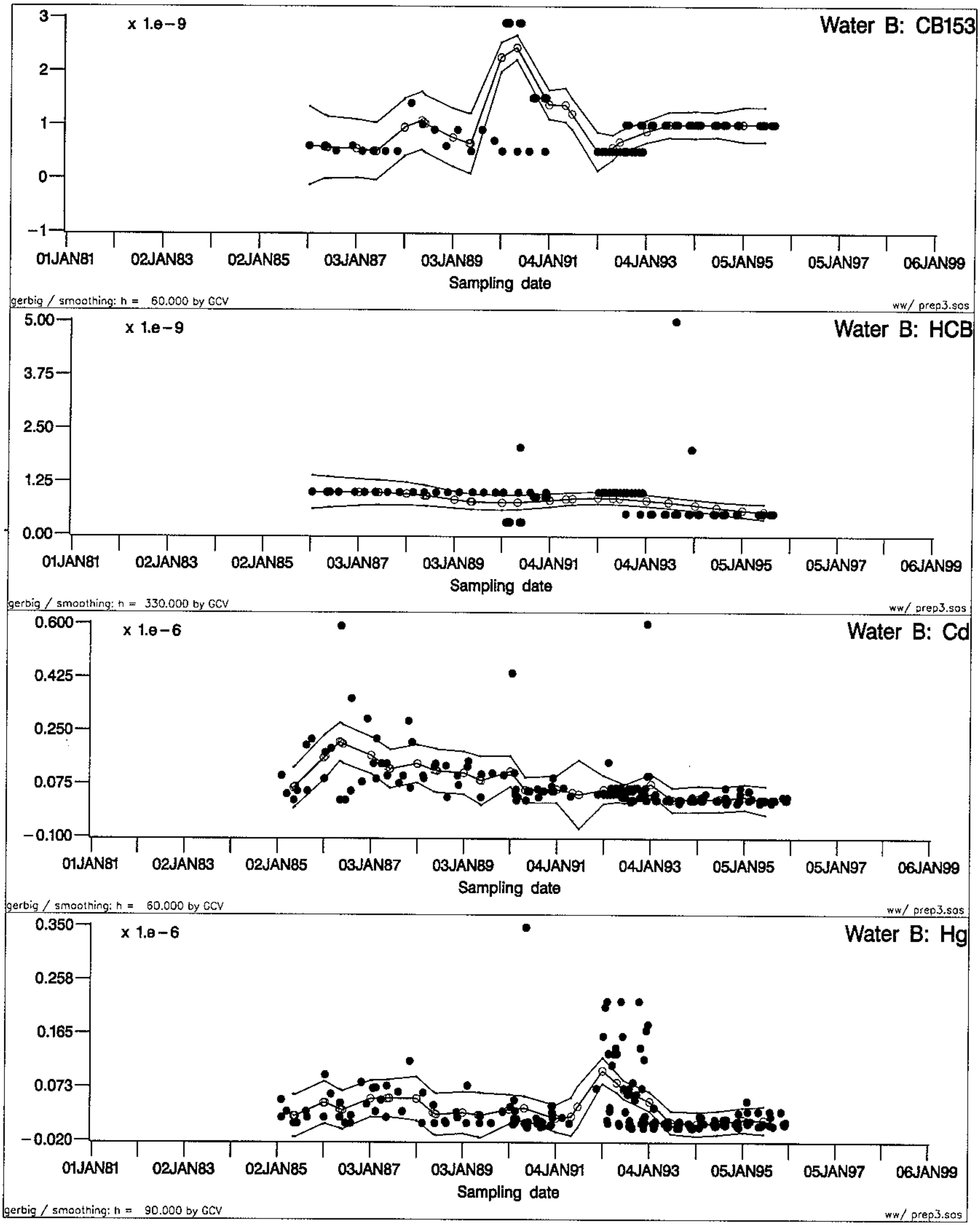


**Figure A5.5.c.** Area 1 (extended German Bight): data on contaminants in sediments (fraction < 63μm) from the ICES Environmental Data Centre (extract) (filled circles: empirical data; empty circles: interpolated values).

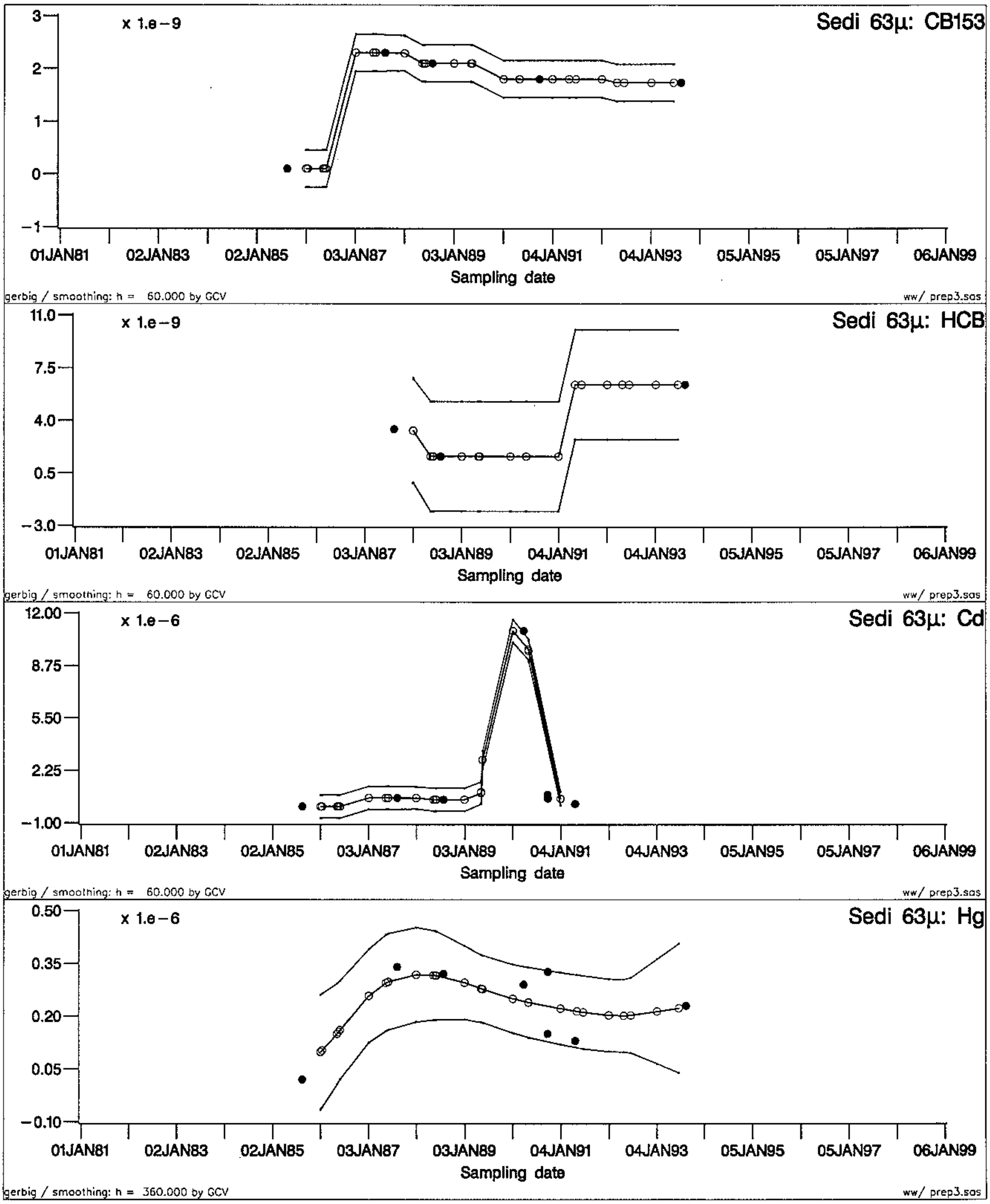


**Figure A5.5.d.** Area 1 (extended German Bight): data on contaminants in *Mytilus edulis* (soft body tissue) from the ICES Environmental Data Centre (extract) (filled circles: empirical data; empty circles: interpolated values).

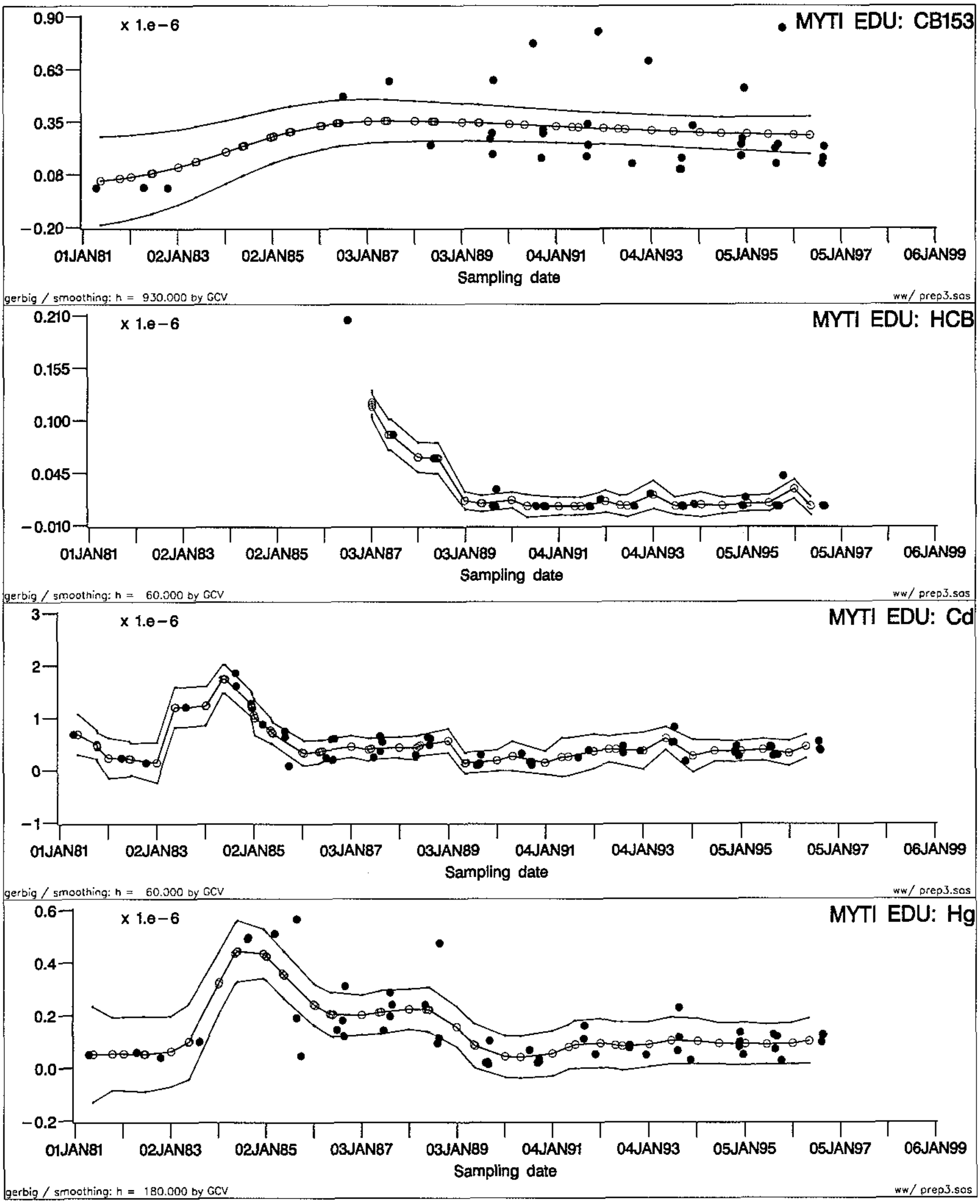


**Figure A5.5.e.** Area 1 (extended German Bight): data on contaminants in liver and muscle of dab (*Limanda limanda*) from the ICES Environmental Data Centre (extract) (filled circles: empirical data; empty circles: interpolated values).

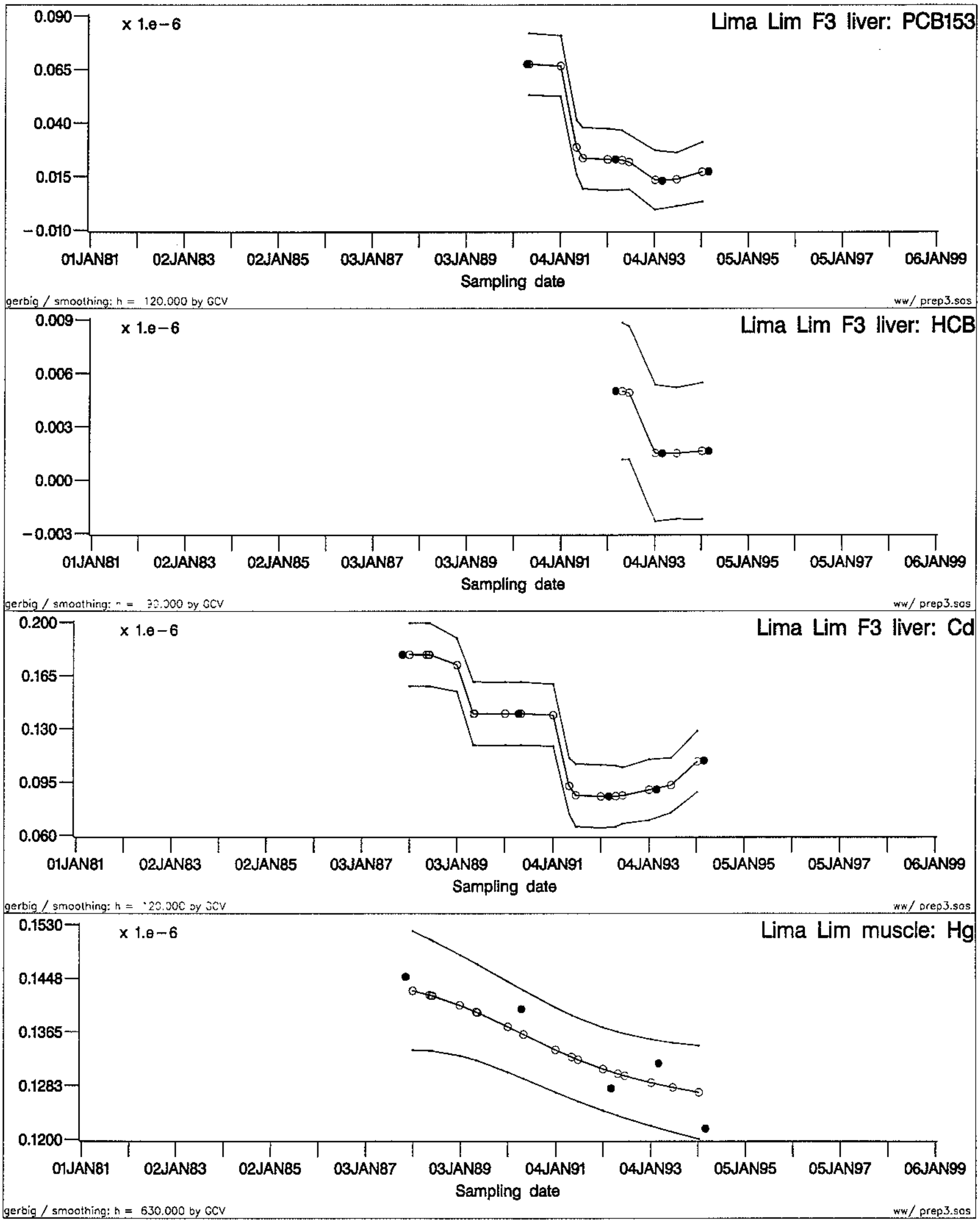
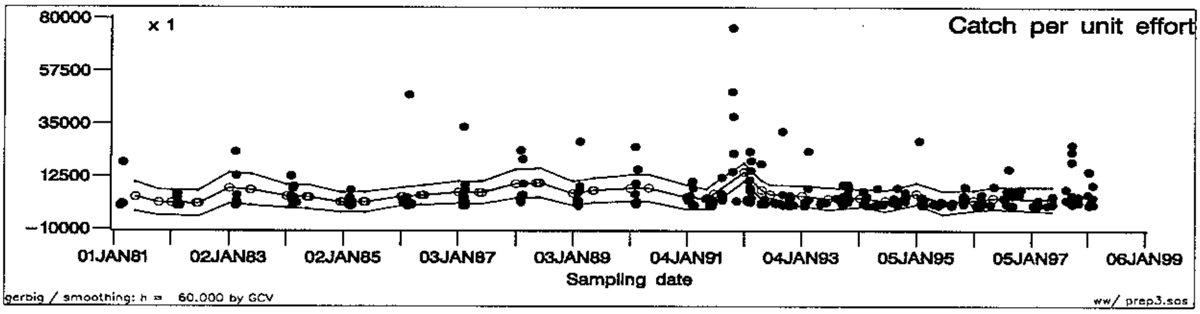
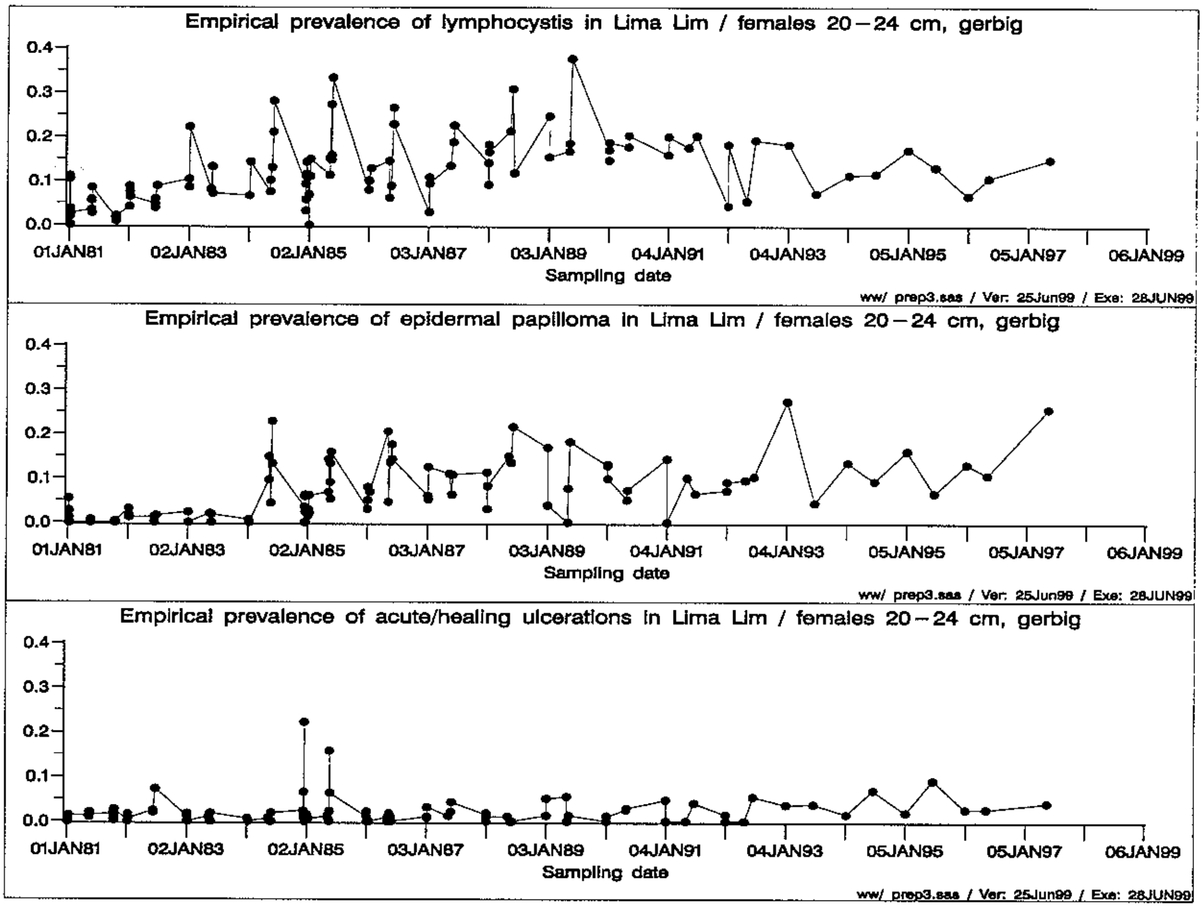


**Figure A5.5.f.** Area 1 (extended German Bight): *Limanda limanda*, data on catch per unit effort (CPUE) (all dab) from the ICES Fishery Data Bank and on diseases (females, 20–24 cm) from the ICES Environmental Data Centre (extract) (filled circles: empirical data; empty circles: interpolated values).



# ANNEX 6

## FOOD CONSUMPTION OF SEABIRDS IN THE ICES AREA, EXCLUDING THE NORTH SEA

## 1 INTRODUCTION

The ICES Working Group on Seabird Ecology (WGSE) constructed a model of food consumption by seabirds (not including seaducks or waders) in the North Sea in 1994 (Tasker and Furness, 1996). This model used information on seabird densities in six sections of the North Sea, along with calculated energy requirements and available information on diet. The total annual seabird energy requirement was $3.9 \times 10^{12}$ kJ, which was the equivalent of 600 000 tonnes of food. The outputs of this model indicate that two species, common guillemot and northern fulmar, were responsible for more than 50 % of the total seabird energy requirements. The energy demand was not homogeneous in space or time—most food was required in ICES Division IVa (west), and the second and third quarters of the year showed the greatest demands. About one-third of the food requirement was met by sandeels, with another one-third deriving from the waste products of fisheries (12 % offal, 18 % discards).

This approach is possible only where there is information on the densities of birds at sea, and on their diet. Such information is available for waters to the west and south of the United Kingdom, in addition to the North Sea. In other areas, numbers of birds at colonies provide some indication of food consumption in the vicinity during the breeding season, but cannot account for immigration or emigration during the non-breeding period. In 1999, the WGSE made calculations of food consumption, based on breeding numbers, for five sections of the Atlantic Ocean outside the North Sea. The North Sea model was also applied to one further area of the sea. The findings of WGSE are reported here.

## 2 STUDIES OFF EASTERN CANADA

Previous models of energy use by seabirds in eastern and Arctic Canada (Diamond *et al.*, 1993; Cairns *et al.*, 1991) allowed estimation of the energy requirement in different oceanographic areas and comparison between these areas. For example, the model of Diamond *et al.* (1993) demonstrated that year-round energy demand by seabirds in the Northwest Atlantic (NWA) was mainly from non-breeding birds, especially populations breeding in the Northeast Atlantic and in the southern hemisphere. However, in the Gulf of St. Lawrence the year-round energy consumption was close to an estimate already based on breeding birds alone (Cairns *et al.*, 1991), illustrating the relatively small impact of the small number of transoceanic and transequatorial migrants in this area (Brown, 1986). At the time that Diamond and colleagues produced their model, dietary information was sparse, both geographically and temporally, and much of it was presented as number of prey items or frequency of occurrence, so neither form allowed a direct prey type harvest assessment for each species of seabird for each oceanographic area. Their approach was mainly aimed at understanding the dynamics of large-scale marine ecosystems. In contrast, the model of Cairns *et al.* (1991) for the Gulf of St. Lawrence provided more specific seabird dietary information from studies within the Gulf and nearby waters. It allowed the estimation of removals of prey type by seabirds in tonnes, a more familiar unit for making comparisons with commercial fisheries landings.

In this section, the breeding populations of the major seabird species in the Gulf of St. Lawrence and Northwest Atlantic, together with data on energy expenditure and diet, provide the basis for estimating food consumption by these populations. For the NWA, transoceanic and transequatorial migrants are also considered in the estimate of energy demands. It must be stressed that the numerous assumptions made in the calculation of prey consumption may be questionable. There are uncertainties regarding the size of the breeding populations, particularly for gull species in the Gulf and NWA. Estimates of the total numbers of seabirds other than breeders are speculative, but reasonable, because they have been based on the population dynamics of the species involved. The residence time or occupation dates by the seabird populations is another parameter liable to introduce errors in the calculation.

### 2.1 Gulf of St. Lawrence (NAFO Areas 4R, 4S, 4T)

The Gulf of St. Lawrence is 214 000 km$^2$ in extent (Steven, 1975) (Figure A6.1). The breeding populations were taken from BIOMQ (Banque Informatisée des Oiseaux Marins du Québec) and from Chapdelaine and Brousseau (1992), Lock *et al.* (1994), and Chapdelaine (1995, 1996). For each species included, the diets assumed for the model are listed in Table A6.1. To estimate numbers of nestlings and pre-breeders, seabirds were classified as inshore or offshore species and the following empirical calculation based on breeding pairs (bp) was adopted for both areas (Cairns *et al.*, 1986; Montevecchi, unpubl.): offshore species = (bp × 0.6) + (bp × 0.8); inshore species = (bp × 0.6) + (bp × 1.0). Approximate occupation dates, population estimates in pairs and number of birds (breeders, nestlings, and non-breeders) identified as the total population using the breeding areas (TPA) are presented in Table A6.2.

Estimates of daily energy expenditure were obtained from measurements of field metabolic rate (FMR) determined by Birt-Friesen *et al.* (1989) or by using allometric equations given by these authors. Estimates of seabird biomass were based on the body-mass values of the birds from this FMR study. In subsequent calculations, it was assumed that the energy requirements

**Figure A6.1.** The NAFO Areas used to describe the Gulf of St. Lawrence and the Northwest Atlantic Ocean.

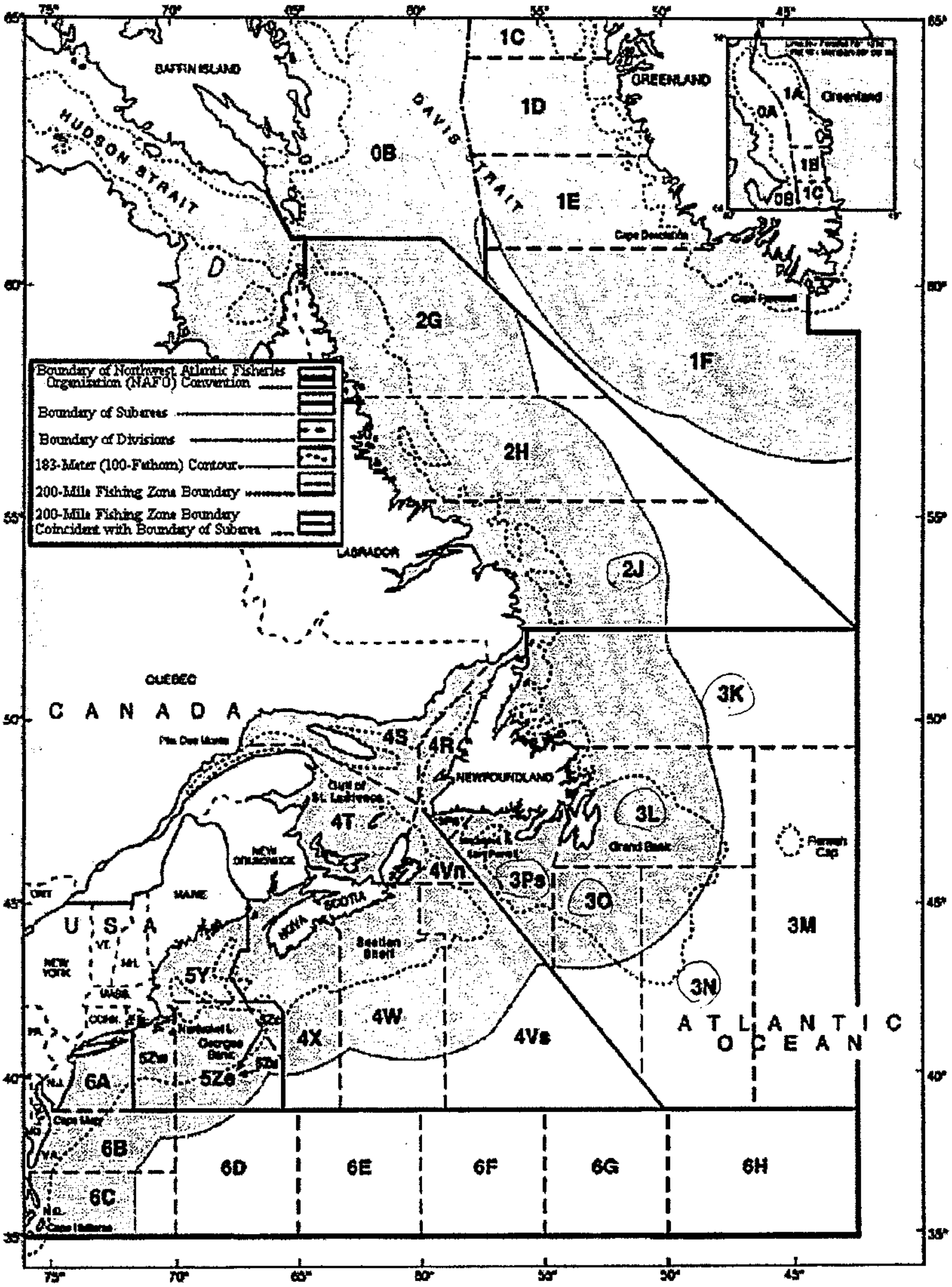


**Table A6.1.** Summary of diets (% mass) for seabirds in the Gulf of St. Lawrence.

| Species | Diets assumed for the model | References |
|---|---|---|
| Leach's storm-petrel | 100 % invertebrates | Montevecchi *et al.*, 1992 |
| Northern gannet | 58 % mackerel, 4 % herring, 10 % capelin, 22 % sandeel, 6 % others | Rail *et al.*, 1996; Burton and Pilon, 1978 |
| Great cormorant | 20 % sandeel, 40 % flatfish, 30 % cunner, 7 % sculpins, 3 % others | Pilon *et al.*, 1983 |
| Double-crested cormorant | 6 % herring, 18 % capelin, 25 % sandeel, 15 % flatfish, 11 % cunner, 10 % sculpins, 15 % others | Pilon *et al.*, 1983; Rail and Chapdelaine, 1998; Gallant, 1988; Léger and Burton, 1979 |
| Black-headed gull | n.a. | n.a. |
| Ring-billed gull | n.a. | n.a. |
| Herring gull | 1 % herring, 58 % capelin, 3 % sandeel, 9 % invertebrates, 29 % others | Rail *et al.*, 1996; Rail *et al.*, in prep. |
| Great black-backed gull | 19 % herring, 57 % capelin, 1 % sandeel, 1 % invertebrates, 1 % sculpins, 21 % others | Rail *et al.*, 1996; Chapdelaine and Rail, unpubl. |
| Black-legged kittiwake | 27 % capelin, 66 % sandeel, 7 % invertebrates | Chapdelaine and Rail, unpubl.; Chapdelaine and Brousseau, 1989 |
| Caspian tern | n.a. | n.a. |
| Common tern | 33 % capelin, 31 % sandeel, 5 % invertebrates, 31 % others | Chapdelaine *et al.*, 1985; Chalifour, 1982 |
| Arctic tern | 17 % capelin, 73 % sandeel, 10 % invertebrates | Chapdelaine *et al.*, 1985 |
| Common guillemot | 97 % capelin, 3 % sandeel | Chapdelaine and Rail, unpubl. |
| Brünnich's guillemot | n.a. | n.a. |
| Razorbill | 58 % capelin, 42 % sandeel | Chapdelaine and Rail, unpubl.; Chapdelaine and Brousseau, 1996 |
| Black guillemot | 8 % sandeel, 1 % invertebrates, 33 % gadidae, 28 % daubed shanny, 30 % others | Cairns, 1981 |
| Atlantic puffin | 37 % capelin, 63 % sandeel | Chapdelaine and Rail, unpubl. |

n.a. = not available

of the birds were stable throughout their respective occupation dates or season. In order to calculate the prey consumption by a seabird, an average energy density of the prey of 6 kJ $g^{-1}$ fresh mass and an assimilation efficiency of 0.75 (Tasker and Furness, 1996) were used.

In order to assess the prey type biomass harvested by seabird species, the dietary information available in the literature related to the Gulf of St. Lawrence was used. Most of it comes from studies at breeding colonies during summer and little is known of the diets outside this period. The partitioning of prey type consumed by seabird is strongly biased toward the summer period and should be interpreted cautiously when these partitions are applied to the entire period in which a species occupies the area. The assessment of seabird diet by numbers of prey items or by frequency of occurrence, as is common in dietary papers, makes it difficult to estimate the biomass of prey consumed by seabirds (Tasker and Furness, 1996). The authors tried to avoid such information in the model, but in some cases this was the only information available. The assumptions made and literature used for input parameters to the model are listed in Table A6.2.

### 2.1.1 Seabird populations

The total number of seabird breeding pairs in the Gulf of St. Lawrence is approximately 370 000 and the total population of birds using the area was estimated at $1.2 \times 10^6$ individuals. The seabird guild is dominated by black-and legged kittiwake (22.5 % of total TPA), but their total biomass represents only 9.2 %. The northern gannet dominates the seabird biomass total with 33.6 % and represents 11.2 % of total TPA. Herring gulls, common guillemots, and double-crested cormorants are the next most important consumers with 13.5 %, 11.4 %, and 11.9 % of total TPA and represent 14.2 %, 11 %, and 16.2 % of the total seabird biomass, respectively.

**Table A6.2.** Seabird species that breed within NAFO Areas 4R, 4S, and 4T in the Gulf of St. Lawrence.

| Species | Calculated population (pairs) | Occupation dates | TPA | Individual mass (kg) | Biomass (kg) | % | FMR (kJ $day^{-1}$) | Ref. FMR data* | Consumption (tonnes $day^{-1}$) | % | Tonnes $year^{-1}$ | % |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Leach's storm-petrel | 518 | May–Oct | 1 761 | 0.05 | 88 | 0.0 | 89 | 1 | 0.0 | 0.0 | 6.4 | 0.0 |
| Northern gannet | 42 124 | Apr–Oct | 143 222 | 3.2 | 458 309 | 33.6 | 4 865 | 1 | 154.8 | 32.8 | 32 779.3 | 30.2 |
| Great cormorant | 2 484 | Apr–Oct | 8 446 | 2.25 | 19 003 | 1.4 | 1 761 | 1b | 3.3 | 0.7 | 699.7 | 0.6 |
| Double-crested cormorant | 39 000 | Apr–Oct | 13 2600 | 1.67 | 221 442 | 16.2 | 1 419 | 1b | 41.8 | 8.9 | 8 851.9 | 8.2 |
| Black-headed gull | 10 | Apr–Oct | 36 | 0.28 | | 0.0 | 473 | 1a | 0.0 | 0.0 | 0.8 | 0.0 |
| Ring-billed gull | 33 392 | Apr–Oct | 120 211 | 0.5 | 60 106 | 4.4 | 1 049 | 1a | 28.0 | 5.9 | 5 932.4 | 5.5 |
| Herring gull | 47 887 | Mar–Dec | 172 393 | 1.12 | 193 080 | 14.2 | 1 984 | 1a | 76.0 | 16.1 | 23 026.1 | 21.2 |
| Great black-backed gull | 9 736 | Mar–Dec | 35 050 | 1.68 | 58 883 | 4.3 | 2 533 | 1a | 19.7 | 4.2 | 5 976.9 | 5.5 |
| Black-legged kittiwake | 84 376 | Apr–Oct | 286 878 | 0.44 | 126 226 | 9.3 | 794 | 1 | 50.6 | 10.7 | 10 715.9 | 9.9 |
| Caspian tern | 11 | May–Sep | 40 | 0.61 | 24 | 0.0 | 1 213 | 1a | 0.0 | 0.0 | 1.6 | 0.0 |
| Common tern | 26 268 | May–Sep | 94 565 | 0.12 | 11 348 | 0.8 | 372 | 1a | 7.8 | 1.7 | 1 169.9 | 1.1 |
| Arctic tern | 1 005 | May–Sep | 3 618 | 0.11 | 398 | 0.0 | 349 | 1a | 0.3 | 0.1 | 42.0 | 0.0 |
| Common guillemot | 4 660 | Apr–Oct | 151 844 | 0.99 | 150 326 | 11.0 | 1 789 | 1 | 60.4 | 12.8 | 12 779.6 | 11.8 |
| Brünnich's guillemot | 12 | Apr–Oct | 41 | 0.93 | 38 | 0.0 | 1 420 | 1 | 0.0 | 0.0 | 2.7 | 0.0 |
| Razorbill | 8 250 | Apr–Oct | 28 050 | 0.72 | 20 196 | 1.5 | 1 368 | 1a | 8.5 | 1.8 | 1 805.2 | 1.7 |
| Black guillemot | 4 762 | Jan–Dec | 16 191 | 0.4 | 6 476 | 0.5 | 616 | 1 | 2.2 | 0.5 | 809.0 | 0.7 |
| Atlantic puffin | 24 174 | Apr–Oct | 82 192 | 0.46 | 37 808 | 2.8 | 988 | 1a | 18.0 | 3.8 | 3 820.3 | 3.5 |
| Total | | | | | | | | | | | 108 419.4 | |

*1 After Birt-Friesen *et al.* (1989)
(1a) cold water flappers FMR= 11.455 $M^{0.727}$ after Birt-Friesen *et al.* (1989), mass in g
(1b) other seabirds FMR= 6.441 $M^{0.727}$ after Birt-Friesen *et al.* (1989), mass in g
(1c) cold water seabirds FMR= 8.892 $M^{0.646}$ after Birt-Friesen *et al.* (1989), mass in g

### 2.1.2 Consumption of food and energy uptake

The northern gannet is the major seabird consumer among the species in the Gulf of St. Lawrence, taking 30 % of the food biomass (Table A6.3). The herring gull, common guillemot, and double-crested cormorant follow with 21 %, 12 %, and 8 % of the food biomass consumed annually by all seabirds. The annual total prey biomass consumed by the breeding population, nestlings, and non-breeders is estimated at 110 000 tonnes. This is the equivalent of $6.5 \times 10^{11}$ kJ of energy required by seabirds in the Gulf or 0.5 g $m^{-2}$ (1 g $m^{-2}$ = 1 t $km^{-2}$ and the entire area is 214 000 $km^2$). Because marine birds are not evenly distributed at sea, the value given for average prey consumption per unit area is not representative of every sector of the Gulf. For their bioenergetics model, Cairns *et al.* (1991) subdivided prey type harvest by NAFO Unit Area for the Gulf of St. Lawrence. They showed that in general the distribution of seabird harvest followed the pattern of breeding colonies.

### 2.1.3 Calculated prey harvest

Fish accounts for 93 % of the prey consumption of this community. Capelin and sandeel are consumed by all seabirds and represent the largest prey components, comprising 36.7 % (39 776 tonnes) and 22.9 % (24 884 tonnes), respectively, of the total consumption (Table A6.3). Larid and alcid species are the most important consumers of these small pelagic and schooling fish. Mackerel is consumed only by northern gannet and accounts for 17.5 % (19 012 tonnes). The total benthic and estuarine fish (mainly represented by flatfish, cunner and sculpins) are consumed mostly by great and double-crested cormorants and comprise 3.9 % of the annual prey harvest by seabirds.

## 2.2 Northwest Atlantic (NAFO Areas 2J, 3K, 3L, 3N, 3O)

The area of the Northwest Atlantic (Figure A6.1) considered in this section is given as 409 766 $km^2$ by Diamond *et al.* (1986). For breeding populations of the NWA, most information is provided by Nettleship and Evans (1985), Montevecchi and Tuck (1987), Nettleship and Chapdelaine (1988, and unpubl.), Cairns *et al.* (1989), Lock *et al.* (1994), Diamond *et al.* (1986, 1993), Sklepkovych and Montevecchi (1989), Stenhouse and Montevecchi (1999), and Montevecchi (unpubl.). The authors used the same assumptions as were used for the Gulf of St. Lawrence to allow for the extra numbers of nestlings and pre-breeders, for FMR, for average energy density of the prey, and for assimilation efficiency. The waters of the open NWA coast are frequented by large numbers of transoceanic and transequatorial migrants, so an estimate of energy use by this group of seabirds was done separately. Information on their numbers is derived from Brown (1986), Diamond *et al.* (1986, 1993), and Montevecchi (unpubl.) (Table A6.4).

### 2.2.1 Seabird populations

The eighteen nesting seabird species within the NAFO Unit Areas 2J, 3K, 3L, 3N, 3O, 3Ps number about 5.6 million pairs. A total population of $1.8 \times 10^7$ individual seabirds was used when assessing food consumption in this area. Most are Leach's storm-petrel (81 %), common guillemot (10 %), and Atlantic puffin (5.2 %) breeding at colonies at Baccalieu Island, Funk Island and Witless Bay Islands (Montevecchi and Tuck, 1987; Cairns *et al.*, 1989; Lock *et al.*, 1994). Gulls and terns (2.7 %), northern gannet (0.3 %), black guillemot, Brünnich's guillemot, razorbill (0.7 %), double-crested cormorant, great cormorant, Manx shearwater and northern fulmar (0.1 %) comprise the remainder of the nesting total. Biomass densities of nesting seabirds in this area represent 9 kg $km^{-2}$ during the breeding season.

In contrast to the Gulf of St. Lawrence, this area supports large numbers of non-breeding migrant seabirds. Their population sizes are poorly known, but the migrants probably outnumber the breeding species here in summer and possibly at all times of the year (Diamond *et al.*, 1993; Lock *et al.*, 1994; Montevecchi, unpubl.). Nine species are considered as occurring within and breeding mostly or completely outside of NAFO Areas 2J, 3K, 3L, 3N, 3O, 3Ps. The Banks offshore of Newfoundland are the chief wintering area for little auks, which represent about 70 % of the migrants group, estimated at 14.3 million birds (Table A6.5). Brünnich's guillemot (10.5 %), great shearwater (10.5 %), black-legged kittiwake (3.5 %), northern fulmar (2.1 %), sooty shearwater (2.1 %), Iceland gull, glaucous gull, and Wilson's storm-petrel (1.3 %) complete the list. Biomass densities of these migrants represent a potential of 12 kg $km^{-2}$ through the year. Thus, the biomass of migrant seabirds exceeds that of the breeders.

### 2.2.2 Consumption of food and energy uptake

The total biomass consumed annually by the breeders (including nestlings and non-breeders) is calculated at 318 351 tonnes. This is the equivalent of an energy consumption of $1.9 \times 10^{12}$ kJ and corresponds to 0.8 g $m^{-2}$ (entire area is 409 766 $km^2$). But this estimate excludes populations breeding in other oceanographic regions present through the year. Guillemot dominates consumption by breeders, with 50.6 % of the total biomass taken in one year. Leach's storm-petrel, Atlantic puffin, herring gull, and northern gannet consume 17.4 %, 15.1 %, 6.4 %, and 3.5 % of the total, respectively (Table A6.5). Northern fulmar, Manx shearwater, great and double-crested cormorants, black-headed, ring-billed and great black-backed gulls, Caspian, common and Arctic terns, Brünnich's guillemot, razorbill and black guillemot comprise the remainder, with a total of 7 %.

**Table A6.3.** Calculated prey harvest by seabirds in the Gulf of St. Lawrence.

| | Pelagic fish | | | | Invertebrates | Benthic and estuarine fish | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Species | Mackerel | Herring | Capelin | Sandeel | | Cunner | Sculpins | Gadidae | Flatfish | Daubed Shanny | Others |
| Leach's storm-petrel | | | | | 6 | | | | | | |
| Northern gannet | 19 012 | 1 311 | 3 278 | 7 211 | | | | | | | 1 967 |
| Great cormorant | | | | 140 | | 210 | 49 | | 280 | | 21 |
| Double-crested cormorant | | 531 | 1 593 | 2 213 | | 974 | 885 | | 1 328 | | 1 328 |
| Herring gull | | 230 | 13 355 | 691 | 2 072 | | | | | | 6 677 |
| Great black-backed gull | | 1 136 | 3 407 | 60 | 60 | | 60 | | | | 1 255 |
| Black-legged kittiwake | | | 2 893 | 7 072 | 750 | | | | | | |
| Common tern | | | 386 | 362 | 59 | | | | | | 363 |
| Arctic tern | | | 7 | 31 | 4 | | | | | | |
| Common guillemot | | | 12 396 | 3 834 | | | | | | | |
| Razorbill | | | 1 047 | 758 | | | | | | | |
| Black guillemot | | | | 65 | 8 | | | 267 | | 226 | 243 |
| Atlantic puffin | | | 1 414 | 2 407 | | | | | | | |
| Total prey harvest | 19 012 | 3 208 | 39 776 | 24 844 | 2 959 | 1 184 | 994 | 267 | 1 608 | 226 | 11 854 |
| % of tonnes/year | 17.5 | 3.0 | 36.7 | 22.9 | 2.7 | 1.1 | 0.9 | 0.2 | 1.5 | 0.2 | 10.9 |

Note: Information did not permit calculations to be made for black-headed gulls, ring-billed gulls, Caspian terns, or Brünnich's guillemot.

**Table A6.4.** Summary of diets (% mass) for seabirds in the Northwest Atlantic Ocean.

| Species | Diets assumed for the model | References |
|---|---|---|
| Northern fulmar | n.a. | n.a. |
| Manx shearwater | n.a. | n.a. |
| Leach's storm-petrel | 100 % invertebrates | Montevecchi *et al.*, 1992 |
| Northern gannet | 41 % mackerel, 14 % herring, 28 % capelin, 2 % sandeel, 10 % saury, 3 % squid, 1 % gadoids, 1 % others | Montevecchi and Myers, 1997 |
| Great cormorant | 2 % invertebrates, 5 % flatfish, 53 % cunner, 12 % sculpins, 1 % gadoids, 27 % others | Milton and Austin-Smith, 1983; Ross, 1976 |
| Double-crested cormorant | 6 % sandeel, 10 % flatfish, 27 % cunner, 16 % sculpins, 41 % others | Milton and Austin-Smith, 1983; Ross, 1976; Lewis, 1957 |
| Black-headed gull | n.a. | n.a. |
| Ring-billed gull | n.a. | n.a. |
| Herring gull | 51 % capelin, 1 % squid, 9 % invertebrates, 2 % gadoids, 37 % others | Threlfall, 1968; Haycock and Threlfall, 1975; Pierotti, 1983; Brown and Nettleship, 1984 |
| Great black-backed gull | n.a. | n.a. |
| Black-legged kittiwake | 75 % capelin, 25 % others | Threlfall, 1968; Regehr, 1994 |
| Caspian tern | n.a. | n.a. |
| Common tern | 33 % capelin, 31 % sandeel, 5 % invertebrates, 31 % others | Chapdelaine *et al.*, 1985 |
| Arctic tern | 17 % capelin, 73 % sandeel, 10 % invertebrates | Chapdelaine *et al.*, 1985 |
| Common guillemot | 86 % capelin, 5 % sandeel, 2 % gadoids, 5 % daubed shanny, 2 % others | Cairns *et al*, 1990; Birkhead and Nettleship, 1983, 1987 |
| Brünnich's guillemot | 29 % capelin, 2 % sandeel, 2 % gadoids, 66 % daubed shanny, 1 % others | Birkhead and Nettleship, 1987 |
| Razorbill | 62 % capelin, 33 % sandeel, 5 % others | Birkhead and Nettleship, 1983; Chapdelaine and Brousseau, 1996 |
| Black guillemot | 8 % sandeel, 1 % invertebrates, 33 % gadoids, 28 % daubed shanny, 30 % others | Cairns, 1981 |
| Atlantic puffin | 81 % capelin, 14 % sandeel, 5 % gadoids | Brown and Nettleship, 1984; Birkhead and Nettleship, 1983; Rodway and Montevecchi, 1996 |

n.a.= not available

The bioenergetics model calculates that the migrants group removes 388 933 tonnes year$^{-1}$ of living prey from the Northwest Atlantic (Table A6.6). Little auk and Brünnich's guillemot take 63.2 %, great shearwater 16.3 %, northern fulmar 9.2 %, black-legged kittiwake 4.8 %, sooty shearwater 3.0 %, and Wilson's storm-petrel, Iceland and glaucous gull together take 3.6 %. The annual energy consumption requirement for migrants is $2.3 \times 10^{12}$ kJ or 0.8 g m$^{-2}$.

Combining the annual consumption of breeders and migrants gives 710 000 tonnes of fish and invertebrates consumed by seabirds in the Northwest Atlantic. This is equivalent to $4.2 \times 10^{12}$ kJ or 1.7 g m$^{-2}$, which is essentially identical to the estimate of Diamond *et al.* (1993) using an energy modelling approach for the same area.

### 2.2.3 Calculated prey harvest

The partitioning of prey type harvest according to the different seabird species shows that capelin is the most important prey consumed by breeders in NWA, with 201 393 tonnes calculated (Table A6.7). It represents 62.7 % of the total annual prey type harvested by seabirds. Common guillemot is the most important predator, with a calculated 138 452 tonnes or 69 % of the capelin taken annually by seabirds. Sandeel represents the second most important type of prey fish but yields only 5.1 % (16 242 tonnes) of the total annual harvest. It is mainly consumed by common guillemot and Atlantic puffin, but its availability does not appear to be the same as in the North Sea, where sandeels constitute the staple food of most of the seabird community (Tasker and Furness, 1996). Mackerel and herring are consumed only by northern gannet and represent merely 1.9 % of the total biomass harvested. Invertebrates are mostly consumed by the abundant Leach's storm-petrel that breeds in the NWA. More specifically, myctophids, amphipods, euphausiids as well as decapods, copepods and isopods constitute their diet, but owing to small body size and metabolic efficiency, they account for comparatively little of the energy that flows through the avian assemblage of the NWA.

**Table A6.5.** Seabird species that breed within NAFO Areas 2J, 3K, 3L, 3N, 3O in the Northwest Atlantic Ocean.

| Species | Calculated population (pairs) | Occupation dates | TPA | Individual mass (kg) | Biomass | % | FMR (kJ $day^{-1}$) | Ref. FMR data* | Consumption (tonnes $day^{-1}$) | % | Tonnes $year^{-1}$ | % |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Northern fulmar | 53 | Jan–Dec | 181 | 0.8 | 145 | 0.0 | 1 477 | 1a | 0.1 | 0.0 | 14.3 | 0.0 |
| Manx shearwater | 100 | Mar–Nov | 340 | 0.48 | 163 | 0.0 | 573 | 1b | 0.0 | 0.0 | 9.2 | 0.0 |
| Leach's storm-petrel | 4 511 952 | Apr–Oct | 15 340 636 | 0.05 | 767 032 | 20.6 | 89 | 1 | 303.4 | 20.1 | 55 371.2 | 17.4 |
| Northern gannet | 14 355 | Apr–Oct | 48 806 | 3.2 | 156 179 | 4.2 | 4 865 | 1 | 52.8 | 3.5 | 11 170.3 | 3.5 |
| Great cormorant | 167 | Mar–Nov | 601 | 2.25 | 1 352 | 0.0 | 1 761 | 1b | 0.2 | 0.0 | 49.8 | 0.0 |
| Double-crested cormorant | 291 | Mar–Nov | 1 048 | 1.67 | 1 750 | 0.0 | 1 419 | 1b | 0.3 | 0.0 | 70.0 | 0.0 |
| Black-headed gull | 7 | Jan–Dec | 25 | 0.28 | 7 | 0.0 | 473 | 1a | 0.0 | 0.0 | 0.6 | 0.0 |
| Ring-billed gull | 6 406 | Apr–Oct | 23 062 | 0.5 | 11 531 | 0.3 | 1 049 | 1a | 5.4 | 0.4 | 1 138.1 | 0.4 |
| Herring gull | 42 163 | Jan–Dec | 151 787 | 1.12 | 170 001 | 4.6 | 1 984 | 1a | 66.9 | 4.4 | 20 273.8 | 6.4 |
| Great black-backed gull | 3 461 | Jan–Dec | 12 460 | 1.68 | 20 933 | 0.6 | 2 533 | 1a | 7.0 | 0.5 | 2 124.8 | 0.7 |
| Black-legged kittiwake | 81 617 | Jan–Dec | 293 822 | 0.44 | 129 282 | 3.5 | 794 | 1 | 51.8 | 3.4 | 10 975.2 | 3.4 |
| Caspian tern | 30 | May–Oct | 108 | 0.61 | 66 | 0.0 | 1 213 | 1a | 0.0 | 0.0 | 4.4 | 0.0 |
| Common tern | 3 091 | Jan–Dec | 11 128 | 0.12 | 1 335 | 0.0 | 372 | 1a | 0.9 | 0.1 | 137.7 | 0.0 |
| Arctic tern | 4 544 | May–Oct | 16 358 | 0.11 | 1 799 | 0.0 | 349 | 1a | 1.3 | 0.1 | 189.9 | 0.1 |
| Common guillemot | 562 605 | Jan–Dec | 1 912 857 | 0.99 | 1 893 728 | 51.0 | 1 789 | 1 | 760.5 | 50.4 | 160 990.8 | 50.6 |
| Brünnich's guillemot | 12 000 | Jan–Dec | 40 800 | 0.93 | 37 944 | 1.0 | 1 420 | 1 | 12.9 | 0.9 | 2 725.6 | 0.9 |
| Razorbill | 10 972 | Jan–Dec | 37 305 | 0.72 | 26 860 | 0.7 | 1 368 | 1a | 11.3 | 0.8 | 2 400.8 | 0.8 |
| Black guillemot | 15 000 | Jan–Dec | 54 000 | 0.4 | 21 600 | 0.6 | 616 | 1 | 7.4 | 0.5 | 2 698.1 | 0.8 |
| Atlantic puffin | 303 781 | Jan–Dec | 1 032 855 | 0.46 | 475 113 | 12.8 | 988 | 1a | 226.8 | 15.0 | 48 007.0 | 15.1 |
| Total | | | | | | | | | | | 318 351.3 | |

*1 After Birt-Friesen *et al.* (1989)
(1a) cold water flappers FMR= 11.455 $M^{0.727}$ after Birt-Friesen *et al.* (1989), mass in g
(1b) other seabirds FMR= 6.441 $M^{0.727}$ after Birt-Friesen *et al.* (1989), mass in g
(1c) cold water seabirds FMR= 8.892 $M^{0.646}$ after Birt-Friesen *et al.* (1989), mass in g

**Table A6.6.** Seabird species that occur within and breed outside of NAFO Areas 2J, 3K, 3L, 3N, and 3O in the Northwest Atlantic Ocean.

| Species | TPA | Individual mass (kg) | Biomass (kg) | % | FMR (kJ $day^{-1}$) | Ref. FMR data* | Tonnes $day^{-1}$ | % | Tonnes $year^{-1}$ | % |
|---|---|---|---|---|---|---|---|---|---|---|
| Wilson's storm-petrel | 50 000 | 0.04 | 2 000 | 0.0 | 119 | 1 | 1.3 | 0.1 | 280 | 0.1 |
| Northern fulmar | 300 000 | 0.8 | 240 000 | 4.7 | 1 477 | 1a | 98.5 | 4.8 | 35 940 | 9.2 |
| Greater shearwater | 1 500 000 | 0.89 | 1 335 000 | 26.2 | 897 | 1b | 299.0 | 14.6 | 63 298 | 16.3 |
| Sooty shearwater | 300 000 | 0.79 | 237 000 | 4.6 | 823 | 1b | 54.9 | 2.7 | 11 615 | 3.0 |
| Iceland gull | 100 000 | 0.86 | 86 000 | 1.7 | 1 557 | 1a | 34.6 | 1.7 | 7 325 | 1.9 |
| Glaucous gull | 50 000 | 1.7 | 85 000 | 1.7 | 2 664 | 1a | 29.6 | 1.4 | 6 266 | 1.6 |
| Black-legged kittiwake | 500 000 | 0.44 | 220 000 | 4.3 | 794 | 1 | 88.2 | 4.3 | 18 677 | 4.8 |
| Brünnich's guillemot | 1 500 000 | 0.93 | 1 395 000 | 27.4 | 1 420 | 1 | 473.3 | 23.1 | 100 205 | 25.8 |
| Little auk | 10 000 000 | 0.15 | 1 500 000 | 29.4 | 437 | 1a | 971.1 | 47.4 | 145 327 | 37.4 |
| Total | | | | | | | | | 388 933 | |

*1 After Birt-Friesen *et al.* (1989)
(1a) cold water flappers FMR= 11.455 $M^{0.727}$ after Birt-Friesen *et al.* (1989), mass in g
(1b) other seabirds FMR= 6.441 $M^{0.727}$ after Birt-Friesen *et al.* (1989), mass in g
(1c) cold water seabirds FMR= 8.892 $M^{0.646}$ after Birt-Friesen *et al.* (1989), mass in g

**Table A6.7.** Calculated prey harvest by seabirds in the Northwest Atlantic Ocean.

| Species | Pelagic fish and squid | | | | | | Invertebrates | Benthic and estuarine fish | | | | | Others |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Mackerel | Herring | Capelin | Sandeel | Atlantic Saury | Squid | | Flatfish | Cunner | Sculpins | Gadoids | Daubed Shanny | |
| Leach's storm-petrel | | | | | | | 55 371 | | | | | | 5 |
| Northern gannet | 4 580 | 1 564 | 3 128 | 223 | 1 117 | 335 | | | | | 111.7 | | 112 |
| Great cormorant | | | | | | | 1 | 2.5 | 26 | 6 | 0.5 | | 13 |
| Double-crested cormorant | | | | 4 | | | | 7.0 | 19 | 11 | | | 29 |
| Herring gull | | | 10 340 | | | 203 | 1 825 | | | | 405.5 | | 7 501 |
| Black-legged kittiwake | | | 8 231 | | | | | | | | | | 2 744 |
| Common tern | | | 45 | 43 | | | 7 | | | | | | 43 |
| Arctic tern | | | 32 | 139 | | | 19 | | | | | | |
| Common guillemot | | | 138 452 | 8 050 | | | | | | | 3 219.8 | 8 050 | 3 220 |
| Brünnich's guillemot | | | 790 | 54 | | | | | | | | 1 799 | 27 |
| Razorbill | | | 1 489 | 792 | | | | | | | | | 120 |
| Black guillemot | | | | 216 | | | 27 | | | | 890.4 | 755 | 809 |
| Atlantic puffin | | | 38 886 | 6 721 | | | | | | | 2 400.4 | | |
| Total prey harvest | 4 580 | 1 564 | 201 393 | 16 242 | 1 117 | 538 | 57 250 | 9.5 | 45 | 17 | 7 028.3 | 16 604 | 14 623 |
| % of tonnes year$^{-1}$ | 1.4 | 0.5 | 62.7 | 5.1 | 0.3 | 0.2 | 17.8 | 0.0 | 0.0 | 0.0 | 2.2 | 5.2 | 4.6 |

Note: Information did not permit calculations to be made for northern fulmar, Manx shearwater, black-headed gull, ring-billed gull, or great black-backed gull.

Migrants have certainly an important impact on pelagic fish species, as the removals by northern fulmar, great and sooty shearwaters, black-legged kittiwake and Brünnich's guillemot represent 67 % of the total seabird removal of this group. They also consume pelagic fish such as capelin in the NWA area (Rice, 1992; Elliot *et al.*, 1990; Montevecchi and Myers, unpublished data). But having no more details of prey type proportions in their diet, no speculations can be made beyond the available information.

# 3 ICELANDIC WATERS (ICES DIVISION Va)

Seabird numbers for Icelandic waters were obtained from the mid ranges of the figures used by Lloyd *et al.* (1991). To estimate numbers of nestlings and pre-breeders, the same empirical calculation as applied in the Northwest Atlantic was adopted. An FMR of 3.9 BMR (basal metabolic rate) (see Tasker and Furness, 1996) was used to assess daily energy expenditure and food consumption was estimated for 90 days, corresponding to the summer period. The estimate was validated by applying the data of Lilliendahl and Solmundsson (1997) in order to estimate the food requirements of six seabird species in Iceland. The discrepancy between the two model outputs was less than 0.01 %, so it is assumed that the results for the 21 species breeding in Iceland are broadly similar to other studies analysing the food consumption by seabirds in other ICES and NAFO oceanographic areas. The total area used by seabirds around Iceland was assumed to be 225 000 $km^2$.

An assessment of prey type biomass harvested by seabird species is available for the six most numerous species found in Icelandic waters (Lilliendahl and Solmundsson, 1997).

## 3.1 Seabird Populations

The 21 species of seabirds nesting in Iceland number about 12.2 million pairs (Table A6.8). Northern fulmar and Atlantic puffin represent 78 % of this total. Common and Brünnich's guillemots are the next most important species and account for 13 % of the seabird breeding population.

## 3.2 Consumption of Food and Energy Uptake

Not unexpectedly, northern fulmar and Atlantic puffin dominate the consumption of the seabird guild in Iceland, accounting for 69 % of the total biomass taken. Common and Brünnich's guillemots are the two next most important consumers, with 22 % of the total food consumed. The total prey biomass consumed in summer by the breeding population, including nestlings and non-breeders, in Iceland is calculated at 986 196 tonnes of fish and invertebrates (Table A6.8). This is equivalent to $4.9 \times 10^{12}$ kJ or 4.4 g $m^{-2}$.

**Table A6.8.** Calculated summer food consumption of all seabird species breeding in Iceland (see assumptions for Iceland).

| Species | Pairs | FMR (kJ $day^{-1}$) | Tonnes $summer^{-1}$ |
|---|---|---|---|
| Northern fulmar | 5 000 000 | 1 005 | 384 413 |
| Manx shearwater | 5 000 | 573 | 219 |
| Leach's storm-petrel | 5 000 | 89 | 34 |
| British storm-petrel | 5 000 | 119 | 46 |
| Northern gannet | 25 000 | 4 865 | 9 304 |
| Shag | 6 600 | 2 882 | 1 541 |
| Great cormorant | 3 000 | 3 467 | 842 |
| Arctic skua | 4 000 | 2 117 | 686 |
| Black-headed gull | 10 000 | 733 | 594 |
| Common gull | 100 | 783 | 6 |
| Herring gull | 10 000 | 1 669 | 1 352 |
| Glaucous gull | 3 500 | 2 760 | 745 |
| Great black-backed gull | 2 500 | 2 710 | 549 |
| Lesser black-backed gull | 10 000 | 1 583 | 1 282 |
| Black-legged kittiwake | 400 000 | 794 | 25 726 |
| Arctic tern | 100 000 | 308 | 2 495 |
| Common guillemot | 1 200 000 | 1 789 | 164 230 |
| Brünnich's guillemot | 450 000 | 1 420 | 48 308 |
| Razorbill | 450 000 | 1 213 | 41 758 |
| Black guillemot | 50 000 | 1 022 | 3 909 |
| Atlantic puffin | 4 500 000 | 866 | 298 121 |
| Total | | | 986 196 |

## 3.3 Consumption of Prey Type and Energy Uptake for Six Seabird Species

The following analysis is based on Lilliendahl and Solmundsson (1997). Atlantic puffin is the major consumer among this group of seabirds, taking 33 % of the calculated 441 700 tonnes of food harvested over the summer period (Table A6.9a). Common guillemot, northern fulmar, and Brünnich's guillemot are the next most important consumers with 23 %, 17 %, and 16 % of the biomass harvested. Black-legged kittiwake and razorbill take 8 % and 5 %, respectively. The total biomass of fish and invertebrates consumed is the equivalent of $2.2 \times 10^{12}$ kJ of energy required by these six seabird species, or 1.96 g $m^{-2}$ (assuming 225 000 $km^2$ for the entire area used by seabirds in Iceland).

**Table A6.9a.** Summer food consumption in tonnes of six seabird species breeding in Icelandic coastal waters (adapted from Lilliendahl and Solmondsson (1997)).

| **Species** | **Population estimate (inds.)** | **FMR (kJ day$^{-1}$)** | **Tonnes summer$^{-1}$** |
|---|---|---|---|
| Northern fulmar | 4 352 000 | 821 | 73 400 |
| Black-legged kittiwake | 1 363 000 | 795 | 19 600 |
| Common guillemot | 2 590 000 | 2 034 | 102 700 |
| Brünnich's guillemot | 1 512 000 | 2 402 | 71 600 |
| Razorbill | 988 000 | 1 245 | 26 500 |
| Atlantic puffin | 7 342 000 | 1 065 | 147 900 |
| Total | | | 441 700 |

### 3.4 Estimated Prey Harvest for Six Seabird Species

Sandeel is the primary prey, constituting 42 % of the total summer food consumption or 184 400 tonnes, while capelin is the second most important with 38 % or 170 700 tonnes (Table A6.9b). Sandeels are mainly eaten by Atlantic puffin, consuming 59 % of the total sandeel take by seabirds, and common guillemot is the most important consumer of capelin, with 40 % of the total biomass consumed in summer by seabirds. Euphausiids are mainly preyed upon by Brunnich's guillemot, which consumes 42 % of the total euphausiids eaten by seabirds; however, capelin remains the main prey of this species.

**Table A6.9b.** Estimated summer food consumption in tonnes of six species breeding in Iceland in 1994 and 1995. Divided among bird species and by major food items (adapted from Lilliendahl and Solmundsson (1997)).

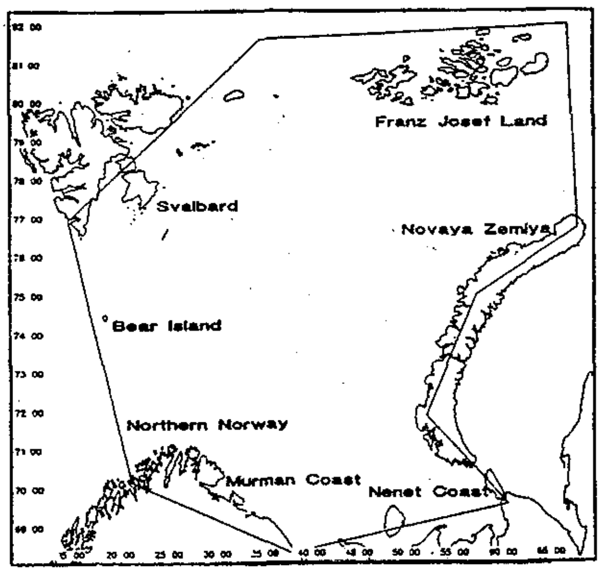
| **Species** | **Capelin** | **Sandeel** | **Euphausiids** | **Others** |
|---|---|---|---|---|
| Northern fulmar | 8 500 | 21 300 | 4 000 | 39 600 |
| Black-legged kittiwake | 15 700 | 3 100 | 400 | 400 |
| Common guillemot | 67 800 | 27 900 | 4 600 | 2 400 |
| Brünnich's guillemot | 41 900 | 10 000 | 14 400 | 5 300 |
| Razorbill | 13 100 | 12 200 | 1 100 | 100 |
| Atlantic puffin | 23 700 | 109 900 | 9 700 | 4 600 |
| Total | 170 700 | 184 400 | 34 200 | 52 400 |

## 4 BARENTS SEA (ICES SUB-AREA I AND EASTERN PARTS OF DIVISIONS IIa, IIb)

This section is based on the work of Mehlum and Gabrielsen (1995). These authors describe the breeding populations of the major seabird species in the Barents Sea region, together with data on energy expenditure and diet. These figures were used to provide the basis for estimating food consumption by these populations and fluxes of energy through the seabirds. In the Barents Sea, year-round energy consumption by seabirds is close to an estimate based on breeding birds alone.

Estimates of breeding population sizes and the assumption that the number of non-breeding adults, chicks and immature birds is equal to the number of breeders were used to calculate the average densities of marine birds. The total Barents Sea area is approximately $1.4 \times 10^6$ km$^2$ (Figure A6.2). Measurements of field metabolic rates of breeding adults during the chick-rearing period with doubly-labelled water were made in several colonies. The mean residency time in the sea was estimated at 250 days of the year. An average energy density and an assimilation efficiency of 5 kJ g$^{-1}$ fresh mass and 0.75 were used, respectively.

**Figure A6.2.** Boundaries of the Barents Sea used in this report.




The estimate of total number of breeding pairs in the Barents Sea region is about $3.7 \times 10^6$, dominated by Brünnich's guillemot. Brünnich's guillemot is also the major consumer, taking 63 % of the food biomass (Table A6.10). The other major species are black-legged kittiwake, common guillemot, Atlantic puffin, and little auk. The annual total prey biomass consumed by the breeding population of marine birds in the entire Barents Sea was calculated at 694 200 tonnes (Table A6.10). Including the non-breeding population and nestlings, the total annual food consumption by birds is estimated at 1 400 000 tonnes. The mean consumption of seabirds in the whole Barents Sea is 1.0 g m$^{-2}$ year$^{-1}$. There are large differences within this huge area and an example of this is that, on a daily basis during the breeding season, the energy flux to the seabirds breeding at Bear Island is five times the average for the whole Barents Sea.

**Table A6.10**. Seabird biomass and food consumption in the Barents Sea.

| Species | Total pairs | Mass (g) | Biomass (kg) | % | FMR (kJ day$^{-1}$) | Consumption (tonnes year$^{-1}$) |
|---|---|---|---|---|---|---|
| Northern fulmar | 27 100 | 650 | 35 230 | 0.8 | 1 005 | 3 625 |
| Black-legged kittiwake | 759 000 | 350 | 561 660 | 13.3 | 788 | 79 750 |
| Glaucous gull | 12 000 | 1 800 | 43 200 | 1.0 | 2 760 | 4 500 |
| Common guillemot | 266 000 | 800 | 425 600 | 10.1 | 1 871 | 66 350 |
| Brünnich's guillemot | 1 567 000 | 820 | 2 569 880 | 60.7 | 2 080 | 434 575 |
| Razorbill | 16 100 | 600 | 19 320 | 0.5 | 1 400 | 3 000 |
| Atlantic puffin | 412 800 | 460 | 379 776 | 9.0 | 848 | 46 650 |
| Black guillemot | 16 200 | 360 | 11 664 | 0.3 | 887 | 1 925 |
| Little auk | 580 000 | 160 | 185 600 | 4.4 | 696 | 53 825 |
| **Total** | | | | | | 694 200 |

## 5 NORWEGIAN SEA (PART OF ICES DIVISION IIA)

### 5.1 Seabird Consumption in the Norwegian Sea

Piscivorous seabirds use the Norwegian Sea (Figure A6.3) for foraging.

**Figure A6.3.** The extent of the Norwegian Sea used in this report (hatched area).

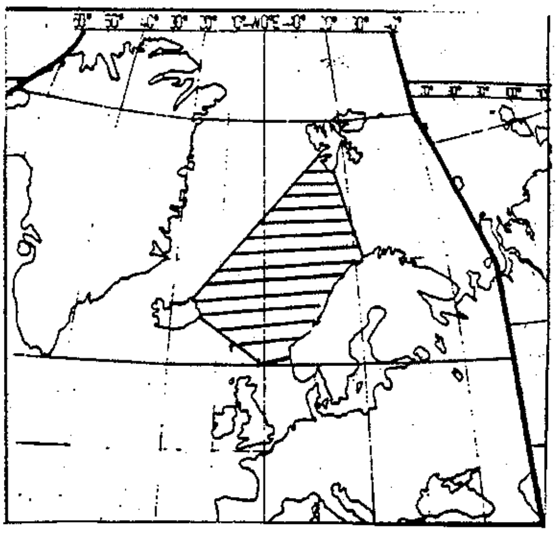


Table A6.11 shows estimates of regional population sizes of breeding seabirds fully or partly dependent on fish prey, taken from Anker-Nilssen (unpublished data) and the Norwegian Seabird Registry at the Norwegian Institute for Nature Research (NINA).

**Table A6.11**. Regional population sizes of breeding seabirds on the Norwegian coastline between Stadt and Lopphavet.

| Species | Population estimates |
|---|---|
| Northern gannet | 5 500 |
| Great cormorant | 30 000 |
| Shag | 25 000 |
| Red-breasted merganser | 30 000 |
| Great skua | 50 |
| Arctic skua | 10 000 |
| Common gull | 150 000 |
| Herring gull | 25 000 |
| Lesser black-backed gull | 3 500 |
| Great black-backed gull | 65 000 |
| Black-legged kittiwake | 325 000 |
| Common tern | 6 000 |
| Arctic tern | 50 000 |
| Razorbill | 30 000 |
| Common guillemot | 10 000 |
| Black guillemot | 30 000 |
| Atlantic puffin | 3 250 000 |

Obviously, among seabirds Atlantic puffins are by far the most numerous of the species, and are the important consumers of fish in this region, constituting 77 % of the seabird numbers (94 % when excluding the more omnivorous gulls and skuas).

Some preliminary calculations for consumption by Atlantic puffins are made in the report by Anker-Nilssen and Øyan (1995). They were based on the following parameters:

1) Daily energy expenditure (DEE) per adult puffin in the chick period: 848 kJ (G.W. Gabrielsen, unpublished data from Hornøy, Finnmark in 1992);

2) DEE outside chick period and by non-breeders assumed reduced by 20 %;

3) Assumed metabolic efficiency of adults: 70 % (Tasker and Furness, 1996);

4) Daily energy demand (DED) per puffin chick: 400 kJ (Øyan and Anker-Nilssen, 1996);

5) Energy value of 0-group herring (mean length 60 mm): 3.7 kJ $g^{-1}$ fresh weight (Anker-Nilssen and Øyan, 1995).

Setting the breeding success to a modest average of 0.6 chick per pair, and extending the calculations to cover 3.25 million breeders and (conservatively) an additional 0.75 million immatures present in the area (as a seasonal average), then the daily energy consumption of Norwegian Sea Atlantic puffins may be calculated as in Table A6.12.

**Table A6.12.** Daily energy consumption of Norwegian Sea Atlantic puffins.

| | |
|---|---|
| Breeders within chick period | 3 937 GJ |
| Nestlings | 390 GJ |
| Breeders outside chick period | 3 150 GJ |
| Non-breeding immatures | 727 GJ |

In total, the Atlantic puffin population in the Norwegian Sea would consume 5054 GJ per day within the chick period, and 3877 GJ per day prior to the chick period. The adults attend their colonies for approximately three months prior to hatching (1 April to 30 June) and another 1.5 months during the chick-rearing period (1 July to 15 August). Given the calculations and assumptions above, the Atlantic puffin's total consumption in this 4.5 month-long breeding season amounts to (3877 × 91) + (5054 × 45) = ca. 580 000 GJ or 156 820 tonnes of herring-equivalent prey (i.e., assuming 3.7 kJ $g^{-1}$, see above).

It is no straightforward task to produce a realistic estimate of the proportion of a herring year-class consumed by these puffins. An attempt to calculate the consumption was made by Anker-Nilssen, Fossum and Gabrielsen (in prep.). A 60-mm long herring (which is the mean size in puffin loads at Røst in good years) weighs about 0.93 g (Anker-Nilssen and Øyan, 1995). Thus, 157 000 tonnes amounts to 157 billion individual herring. Although the puffins also feed on several other prey, the actual 'predation pressure' on herring may be many times larger. This is because the average herring (in good years) grows from less than 0.1 g in April to 5–7 g in July/August. The energy value of herring increases with fish size: 3.5 kJ $g^{-1}$ for 41–50 mm fish, 3.6 kJ $g^{-1}$ for 51–60 mm, 3.8 kJ $g^{-1}$ for 61–70 mm, 4.4 kJ $g^{-1}$ for 71–80 mm, 4.8 kJ $g^{-1}$ for 81–90 mm and 4.9 kJ $g^{-1}$ for 91–100 mm (Anker-Nilssen and Øyan, 1995), but these data suggest that it is relatively constant (approximately 3.4–3.7 kJ $g^{-1}$) for fish in the normal size range available to puffins in the Norwegian Sea during the breeding season (average usually less than 60 mm).

## 6 DISCUSSION

Capelin has a prominent position in the ecosystem of the Gulf of St. Lawrence as a prey species. Many species of fish, marine mammals, and seabirds are dependent on capelin for their survival. Commercial fishery removes only a small proportion of the total biomass because of fluctuating market demand. Prior to 1977, annual landings were stable at under 2000 tonnes. The emergence of a Japanese market for roe-bearing females has attracted the attention of Canadian fishers. Japanese demand is responsible for the sharp increase in landings, which stood at about 10 000 tonnes in 1978 and 1979 and also between 1989 and 1993. Capelin landings totalled 6786 tonnes and 7451 tonnes in 1996 and 1997, respectively (Anon., 1998a). So, it is widely recognized that the fishery in NAFO Areas 4R, 4S, 4T removes only a small portion of the total biomass, compared to cod in the northern Gulf, as well as many other species like seals and summer visitors such as whales and seabirds (Anon., 1998b).

In the Northwest Atlantic, breeding seabirds annually take an estimated 200 000 tonnes of capelin, which is much more important than the consumption in the Gulf. But this harvesting by seabirds is quite small compared to the mass of capelin taken by the main predatory fish and mammals in the Northwest Atlantic. Harp seals are estimated to have consumed about 800 000 tonnes of capelin in NAFO Areas 2J, 3K, 3L in 1996 (Anon., 1998c). Also, previous estimates for cod consumption of capelin indicated that during the early 1980s, cod were consuming 1 million to 3 million tonnes of capelin annually. During the same time period, a minimum of 100 000–200 000 tonnes of capelin were estimated to have been consumed by Greenland halibut (Anon., 1998c). Annual harvest by the fisheries is estimated at about 25 000 tonnes annually (Anon., 1998c).

Capelin is also an important prey in Icelandic waters, but quantities consumed in summer are equalled in order of magnitude by the take of sandeels. Given that sandeels are unavailable in winter, capelin is probably the principle year-round prey. Food consumption in the Norwegian and Barents Seas has not been fully partitioned by prey species, but capelin is likely to be important in both systems.

## 7 ACKNOWLEDGEMENT

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**Scientific names of species (in taxonomic order) mentioned in Annex 6.**

| **Common name** | ***Scientific name*** |
|---|---|
| **Birds** | |
| Northern fulmar | *Fulmarus glacialis* |
| Great shearwater | *Puffinus gravis* |
| Sooty shearwater | *Puffinus griseus* |
| Manx shearwater | *Puffinus puffinus* |
| Wilson's storm-petrel | *Oceanites oceanicus* |
| British storm-petrel | *Hydrobates pelagicus* |
| Leach's storm-petrel | *Oceanodroma leucorhoa* |
| Northern gannet | *Morus bassanus* |
| Double-crested cormorant | *Phalacrocorax auritus* |
| Great cormorant | *Phalacrocorax carbo* |
| Shag | *Phalacrocorax aristotelis* |
| Red-breasted merganser | *Mergus serrator* |
| Common gull | *Larus canus* |
| Ring-billed gull | *Larus delawarensis* |
| Great black-backed gull | *Larus marinus* |
| Glaucus gull | *Larus hyperboreus* |
| Iceland gull | *Larus glaucoides* |
| Herring gull | *Larus argentatus* |
| Lesser black-backed gull | *Larus fuscus* |
| Black-headed gull | *Larus ridibundus* |
| Black-legged kittiwake | *Rissa tridactyla* |
| Great skua | *Catharacta skua* |
| Arctic skua | *Stercorarius parasiticus* |
| Caspian tern | *Sterna caspia* |
| Roseate tern | *Sterna dougallii* |
| Common tern | *Sterna hirundo* |
| Arctic tern | *Sterna paradisaea* |
| Little auk | *Alle alle* |
| Guillemot | *Uria aalge* |
| Brünnich's guillemot | *Uria lomvia* |
| Razorbill | *Alca torda* |
| Black guillemot | *Cepphus grylle* |
| Atlantic puffin | *Fratercula arctica* |
| **Fish** | |
| Atlantic herring | *Clupea harengus* |
| Capelin | *Mallotus villosus* |
| Atlantic cod | *Gadus morhua* |
| Cunner | *Tautogolabrus adspersus* |
| Sandeel (sand lance) | *Ammodytes* spp. |
| Atlantic mackerel | *Scomber scombrus* |
| Atlantic saury | *Scomberesox saurus* |
| Sculpin | *Myoxocephalus* sp. |
| Daubed shanny | *Sticheeus* sp. |
| Greenland halibut | *Reinhardtius hippoglossoides* |
| **Invertebrates** | |
| Squid | *Illex illecebrosus* |
| **Marine Mammals** | |
| Harp seal | *Phoca groenlandica* |

# ANNEX 7

# SEABIRDS AS MONITORS OF MARINE CONTAMINATION

## 1 INTRODUCTION

The use of seabirds as monitors of marine pollution has been advocated many times (Chapdelaine *et al.*, 1987; Furness, 1987, 1993; Gilbertson *et al.*, 1987; Becker, 1989, 1991; Walsh, 1990; Furness *et al.*, 1995; Monteiro and Furness, 1995; Barrett *et al.*, 1996; Elliott *et al.*, 1996; Becker *et al.*, 1998). In this review, the usefulness of seabirds as a means of monitoring contaminants in marine ecosystems is considered. Particular situations where the monitoring of contaminants in seabirds is highly desirable as a cost-effective and informative procedure are described.

Seabird eggs are useful in contaminant monitoring because eggs offer a stable, well-defined matrix with a high fat content, where many contaminants, especially organic substances, accumulate (ICES, 1995). Thus, eggs can be used for detecting spatial as well as temporal variations. Also, as birds are top predators, with a high rate of bioaccumulation, the monitoring of contaminants in birds can be combined with studies of biological effects on populations. Potential choices of seabird species in the Northeast Atlantic and Baltic Sea are given in ICES (1995). Some examples of estimates of variance components in the analysis of several contaminants in five species of seabirds are given in ICES (1999).

## 2 SEABIRDS AS BIOMONITORS OF MARINE CONTAMINATION

### 2.1 Oil Pollution

The use of seabirds as monitors of oil pollution at sea has been reviewed in a number of recent publications (Camphuysen and van Franeker, 1992; Dahlmann *et al.*, 1994; Camphuysen, 1995, 1998; Wiens *et al.*, 1996; Furness and Camphuysen, 1997). Beached bird surveys carried out predominantly by amateurs with organisation and data interpretation by professional staff (usually from non-governmental organizations (NGOs)) provide clear evidence of long-term trends in oiling rates of seabirds (Figure A7.1) and differences in oil impacts among regions (Figure A7.2). There is evidence to show that oiling indices based on the proportion of beached seabirds with oil give a reasonable measure of the number of oil slicks at sea, although factors such as wind direction and numbers of seabirds dying from starvation or disease can confound the picture (Stowe, 1982). Recent developments in fingerprinting oil from carcasses permit identification of the source of oil on birds and can be used in prosecutions for the discharge of oil at sea (Dahlmann *et al.*, 1994). Toxic effects of oil ingested by seabirds have been reviewed several times (e.g., Briggs *et al.*, 1996), but are unlikely to provide a useful monitor of oil impact on seabirds. Seabirds as monitors of oil pollution are not considered further in this review, but readers interested in this topic are referred to the papers listed above.

### 2.2 Plastic Particle Pollution

The use of seabirds as monitors of plastic particle pollution on the ocean surface has been suggested by a number of authors (Furness, 1985, 1993; Ryan, 1987; Spear *et al.*, 1995; Blight and Burger, 1997) because some seabirds, especially petrels, accumulate large numbers of plastic particles in their gizzard, and may suffer harmful effects in their ability to process food (Ryan, 1988). Sampling seabirds to measure quantities of ingested plastic requires obtaining dead birds or killing birds since the stomach contents must be obtained. Procedures that offload the proventriculus contents by 'stomach-pumping' or 'wet-offloading' do not extract material from the gizzard, which is where the vast majority of plastic is stored. Thus, sampling proventriculus contents from live seabirds provides very little data on plastic ingestion and is unlikely to be an effective means of monitoring plastic ingestion. Killing seabirds as a means of monitoring pollution is undesirable. However, there are two possible sources of seabird gizzards that do not require the killing of healthy birds. It may be possible to monitor plastic ingestion by sampling from petrels obtained from beached bird surveys (e.g., fulmars *Fulmarus glacialis* on the European beaches). However, this runs the risk that the birds washed up on beaches do not represent the population as a whole. Beached birds are likely to be predominantly juveniles rather than adults, and birds that have died slowly may have ingested plastic more or less than birds that are healthy and feeding well. Such biases may be difficult to quantify. Another possible source of gizzards from petrels may be from colonies where birds are killed in numbers by predators or accidents. At some colonies, petrels form the main prey of skuas and gizzards may be obtained from some of the birds killed by skuas before the skuas have eaten them. At other colonies, birds may be killed by cats or rats or due to specific hazards. For example, shearwater fledglings at some colonies die when attracted to lights; prions at some colonies become tangled in vegetation. Sampling on a regular basis over years at such sites might provide indications of long-term trends in plastic particle pollution. There is evidence that plastic particle burdens in pelagic seabirds are increasing and that this is a problem that needs further attention but this topic will also not be considered further here, since it is assumed that the main focus of interest is in contamination by chemicals.

**Figure A7.1.** Trends in oil rates of razorbills, guillemots, kittiwakes, and *Larus* gulls stranded at the mainland coast in the Netherlands, 1979–1995. Data from Camphuysen (1995).

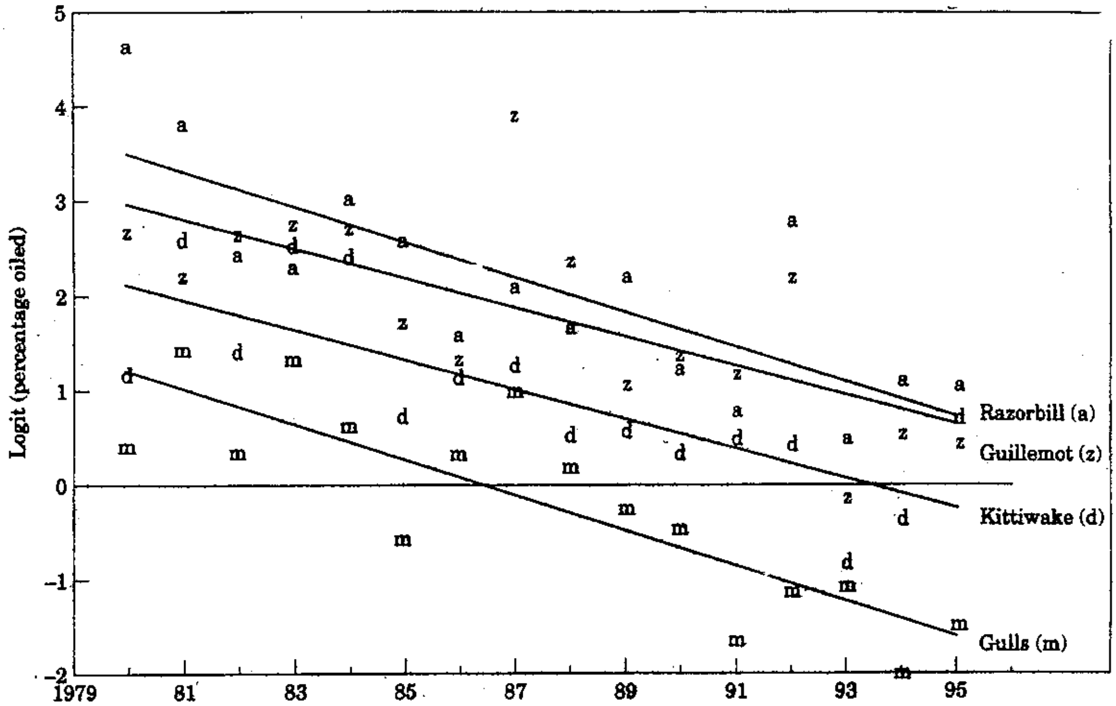
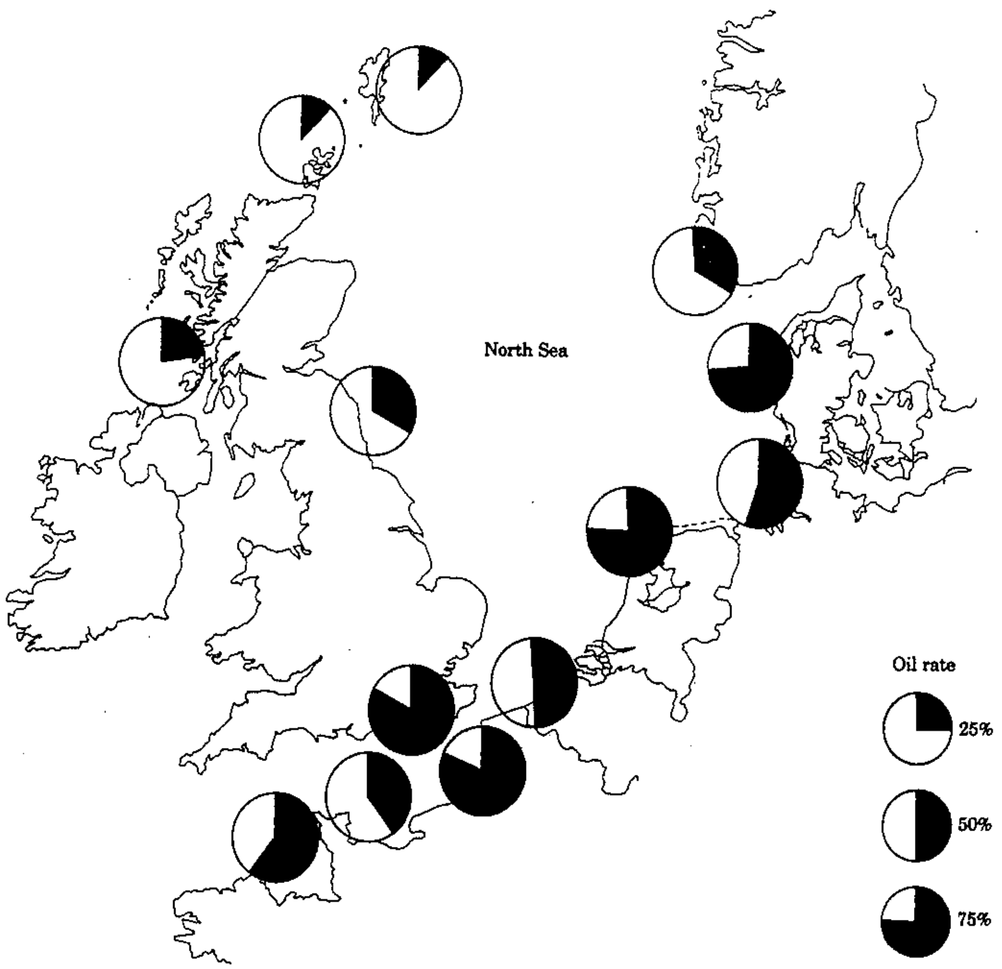



**Figure A7.2.** Oil rates of guillemots at various locations in western Europe. Data from Camphuysen (1995).



## 2.3 Organochlorines

Organochlorine concentrations have been measured in the physical environment (e.g., Bignert *et al.*, 1998), in marine invertebrates (e.g., Johansen *et al.*, 1996; Mattig *et al.*, 1997), in fish (e.g., von Westernhagen, 1994; Jones and Franklin, 1997; Kennish and Ruppel, 1998; Mattig *et al.*, 1997), and in various populations of marine mammals (e.g., Addison *et al.*, 1984; Jarman *et al.*, 1997) and seabirds (e.g., Mehlum and Daelemans, 1995; Savinova *et al.*, 1995; Focardi *et al.*, 1996; Jones *et al.*, 1996; Joiris *et al.*, 1997; van den Brink, 1997; van den Brink *et al.*, 1997).

Being lipid-soluble, organochlorines tend to accumulate in the lipid-rich tissues of animals, and biomagnify through the food chain, so that animals high in marine food chains tend to carry the largest body burdens and have the highest tissue concentrations (Figure A7.3, Tables A7.1 and A7.2). The variation of organochlorines within seabird samples, however, is similar or even lower compared with that of their food (Table A7.2).

**Table A7.1.** Mean concentrations (µg $g^{-1}$ wet weight) and coefficients of variation for DDE and PCBs in marine organisms.

| Species | Site | Age | Tissue | n | PCBs | | DDE | | Ref. |
|---|---|---|---|---|---|---|---|---|---|
| | | | | | mean | CV | mean | CV | |
| Herring | Nova Scotia | 4 years | Muscle | 29 | 0.25 | 108 % | 0.06 | 490 % | 1 |
| Herring | Gulf of St. Lawrence | – | Muscle | 26 | 0.44 | 98 % | 0.09 | 506 % | 1 |
| Herring | Wadden Sea | 2–3 years | Whole | 7 | 0.03 | 25 % | 0.005[1)] | 29 % | 2 |
| Sandeel | Wadden Sea | 2–3 years | Whole | 8 | 0.04 | 40 % | 0.002[1)] | 83 % | 2 |
| Plaice | Wadden Sea | 2 years | Whole | 7 | 0.02 | 30 % | 0.002[1)] | 28 % | 2 |
| Flounder | Norway | – | Liver | 10 | 0.03 | 63 % | 0.06 | 116 % | 1 |
| Cod | Norway | – | Liver | 18 | 0.45 | 57 % | 0.70 | 79 % | 1 |
| Cod | Nova Scotia | 5 years | Liver | 38 | 1.81 | 49 % | 0.28 | 50 % | 1 |
| Cod | Nova Scotia | 2–9 yrs | Liver | 100 | 1.71 | 53 % | 0.28 | 54 % | 1 |
| Grey seal | Nova Scotia | immature | Blubber | 8 | 5.00 | 70 % | 1.50 | 40 % | 1 |
| Grey seal | Nova Scotia | adult | Blubber | 8 | 15.7 | 37 % | 2.50 | 32 % | 1 |
| Leach's storm petrel | Newfoundland, 1984 | eggs | Egg | 5 | 1.16 | 24 % | 0.40 | 28 % | 1 |
| Leach's storm petrel | Bay of Fundy, 1984 | eggs | Egg | 5 | 3.44 | 36 % | 1.05 | 39 % | 1 |
| Puffin | Newfoundland, 1984 | eggs | Egg | 5 | 0.99 | 12 % | 0.30 | 19 % | 1 |
| Puffin | Bay of Fundy, 1984 | eggs | Egg | 5 | 3.20 | 20 % | 0.74 | 24 % | 1 |
| Common tern | Wadden Sea | eggs | Egg | 10 | 4.65 | 33 % | 0.39 | 60 % | 3 |
| Herring gull | Wadden Sea | eggs | Egg | 10 | 1.45 | 40 % | 0.39 | 60 % | 3 |
| Oystercatcher | Wadden Sea | eggs | Egg | 10 | 2.66 | 48 % | 0.11 | 46 % | 3 |

[1)] ΣDDT; Ref. = Reference; 1 = Gilbertson *et al.* (1987); 2 = Mattig *et al.* (1996) (year 1992); 3 = Becker *et al.* (1991) (year 1987: Mellum, Minsener, and Oldeoog islands).

**Figure A7.3.** Biomagnification of PCBs in the food web of the Wadden Sea. Sums of the concentrations of eight PCB congeners on a fat weight basis are presented for plankton, nine benthic invertebrate species, four fish (juvenile stages) and three seabird species. Means + 1 standard deviation (Mattig *et al.*, 1996).

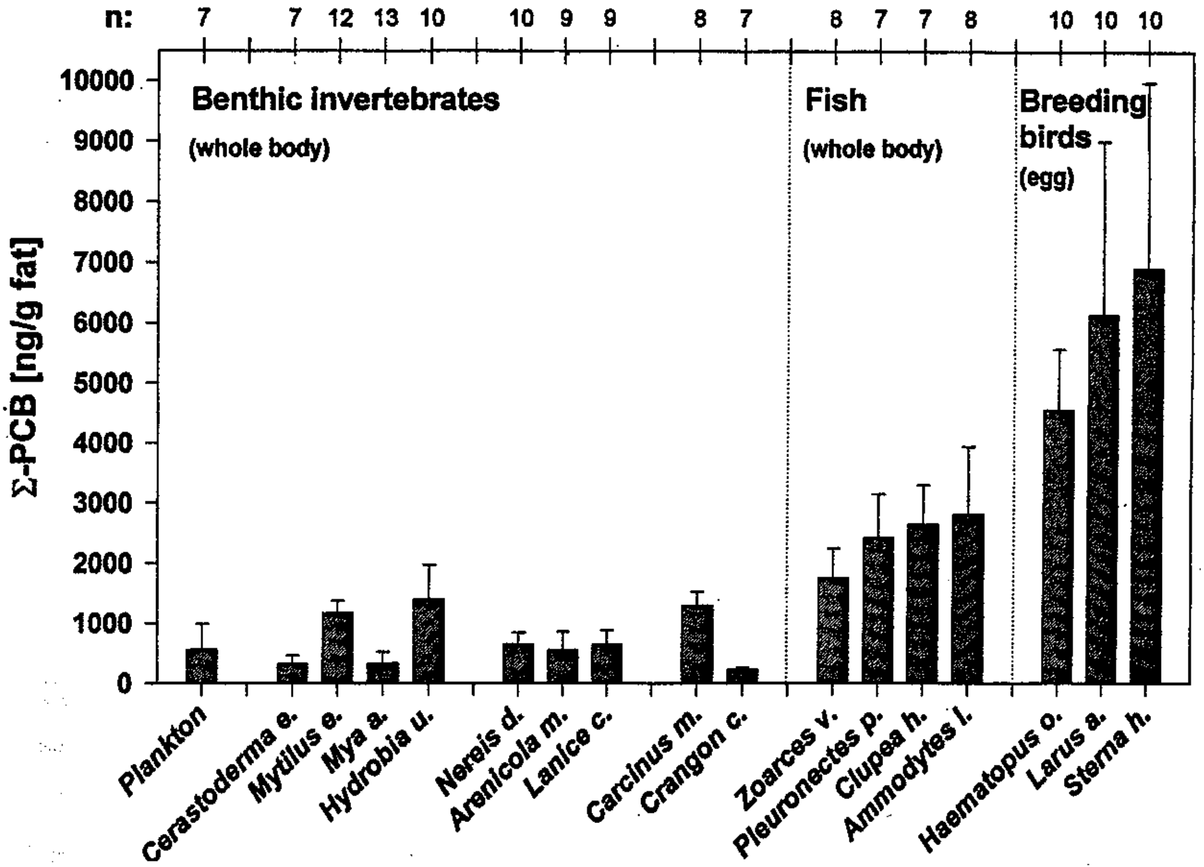



**Table A7.2.** Biomagnification factors between organochlorines in food and eggs of oystercatcher, herring gull and common tern from Spiekeroog, German Wadden Sea in 1993 (Mattig *et al.*, 1996). See Figure A7.3 for PCB concentrations.

| Biomagnification factors | | | | |
|---|---|---|---|---|
| **Species** | **Food** | **PCBs** | **DDTs** | **HCB** |
| Oystercatcher | Benthic animals[1] | 4–14 | 3–23 | 12–21 |
| Herring gull | Benthic animals[1] | 5–19 | 6–46 | 17–30 |
| | Fish [2] | 2–3 | 2 | 2–3 |
| Common tern | Fish [2] | 3 | 3 | 3–5 |

[1] *Cardium, Mytilus*; [2] Herring, plaice.

PCBs are industrial chemicals and consist of 209 congeners. The degree of metabolization of the PCB mixture in seabirds depends on the species, on the level of PCB contamination in the environment, as well as on the length of the time during the year the birds spend in that environment (Beyerbach *et al.*, 1993; Denker *et al.*, 1994).

Different organochlorines, and even different congeners of PCBs, differ considerably in toxicity (Niimi, 1996) and toxic effects vary considerably between different groups of animals. As a broad generalization, seabirds tend to be less sensitive to organochlorine toxicity than marine mammals or terrestrial birds (Beyer *et al.*, 1996). Almost all sampling of adult birds and mammals for monitoring organochlorine concentrations has used samples of liver or muscle tissues, so that animals had to be killed or samples taken opportunistically from drowned or wrecked birds or mammals. These latter samples may have introduced bias due to starvation and consequent mobilization of lipid reserves and, thus, high concentrations of organochlorines. High tissue (especially liver) concentrations of organochlorines can indicate tissue wastage rather than large intakes of the contaminants.

Seabird eggs have been sampled to provide more reliable monitoring of organochlorines, and this permits an assessment of geographical patterns of organochlorine contamination (Becker *et al.*, 1998) (Figure A7.4) as well as long-term trend analysis (Chapdelaine *et al.*, 1987; Elliott *et al.*, 1988, 1996; Bignert *et al.*, 1995; Becker *et al.*, 1998) (Figures A7.5–A7.7). The change in load of

**Figure A7.4.** Geographical patterns of mercury and organochlorine contamination of common tern eggs in the Wadden Sea in 1996 and 1997 (six sites; from Becker *et al.*, 1998). Mean values ± 95% coefficients of variation, n = 10 eggs each.

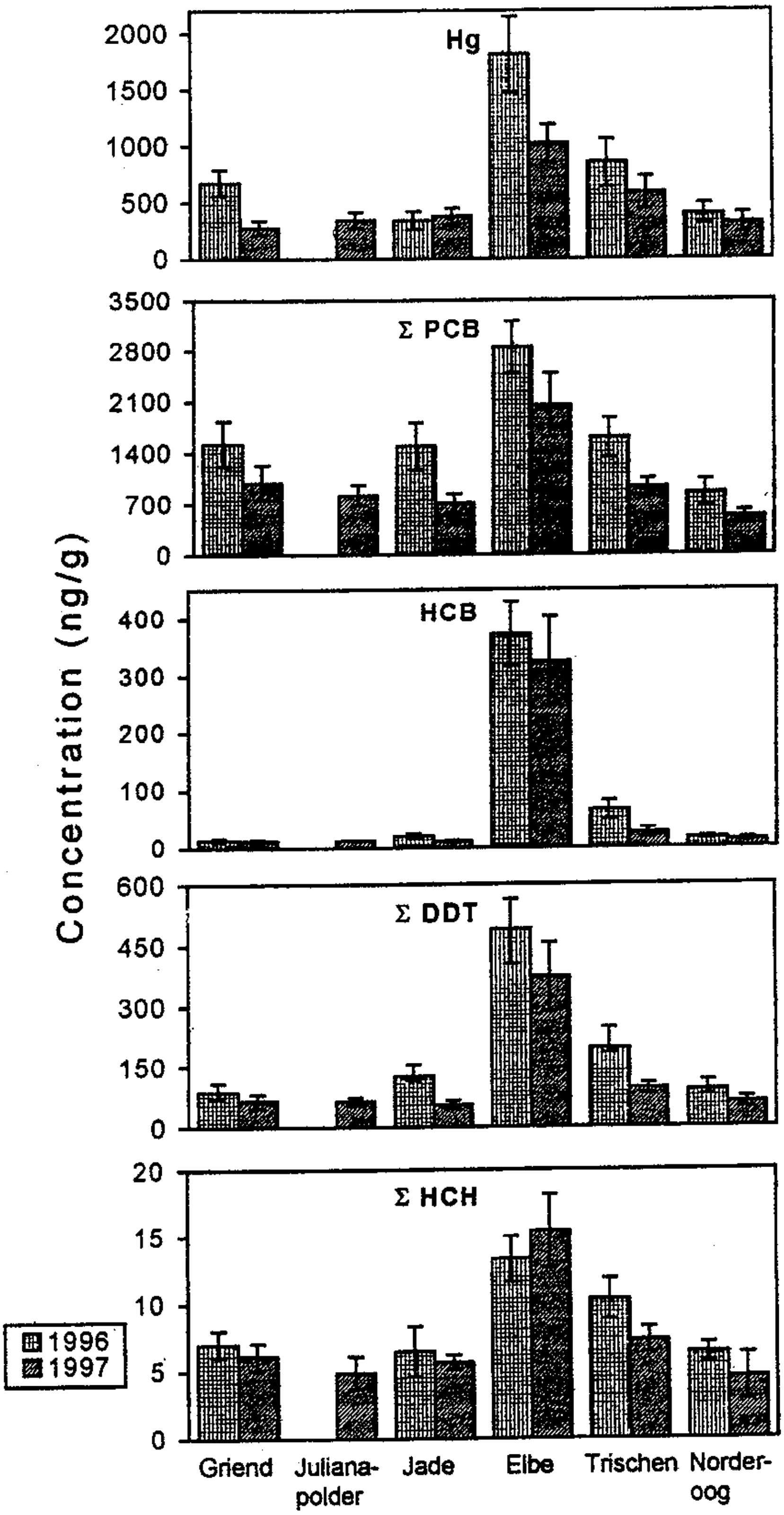


**Figure A7.5.** Temporal trends of PCB concentrations in eggs of common terns from three selected breeding sites in the Wadden Sea from 1981–1997 (Becker *et al.*, 1998).

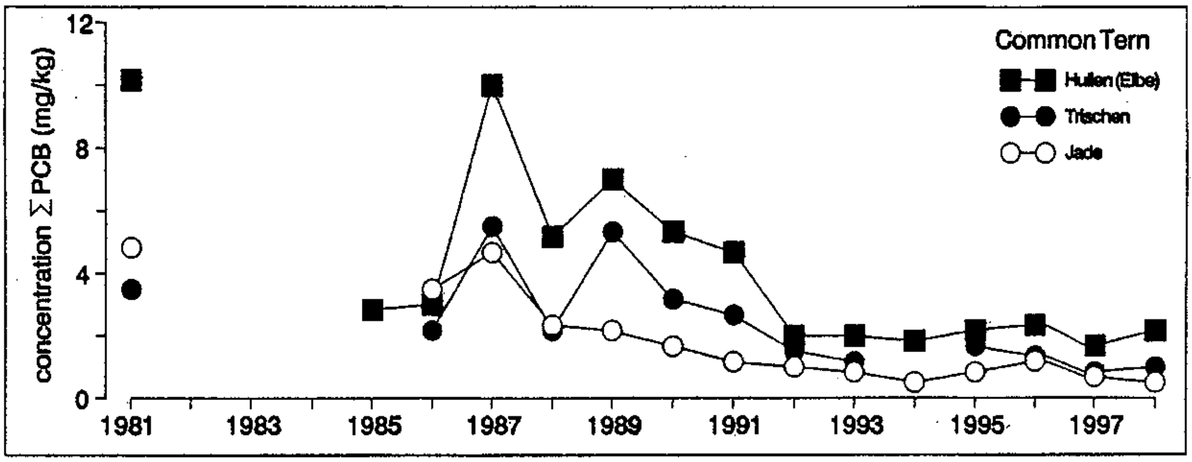



**Figure A7.6.** Temporal trend in PCB loads of the Elbe River compared to the trends in PCB concentrations in eggs of common terns (black columns) and oystercatchers (white columns) breeding at the Elbe estuary (Bakker *et al.*, 1997).

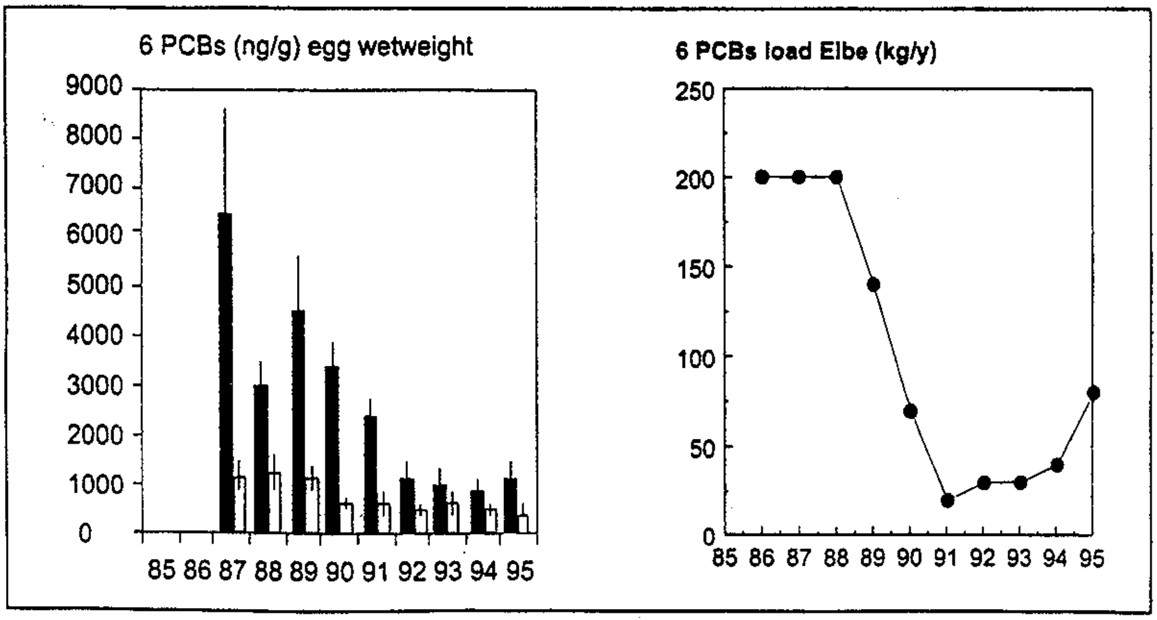
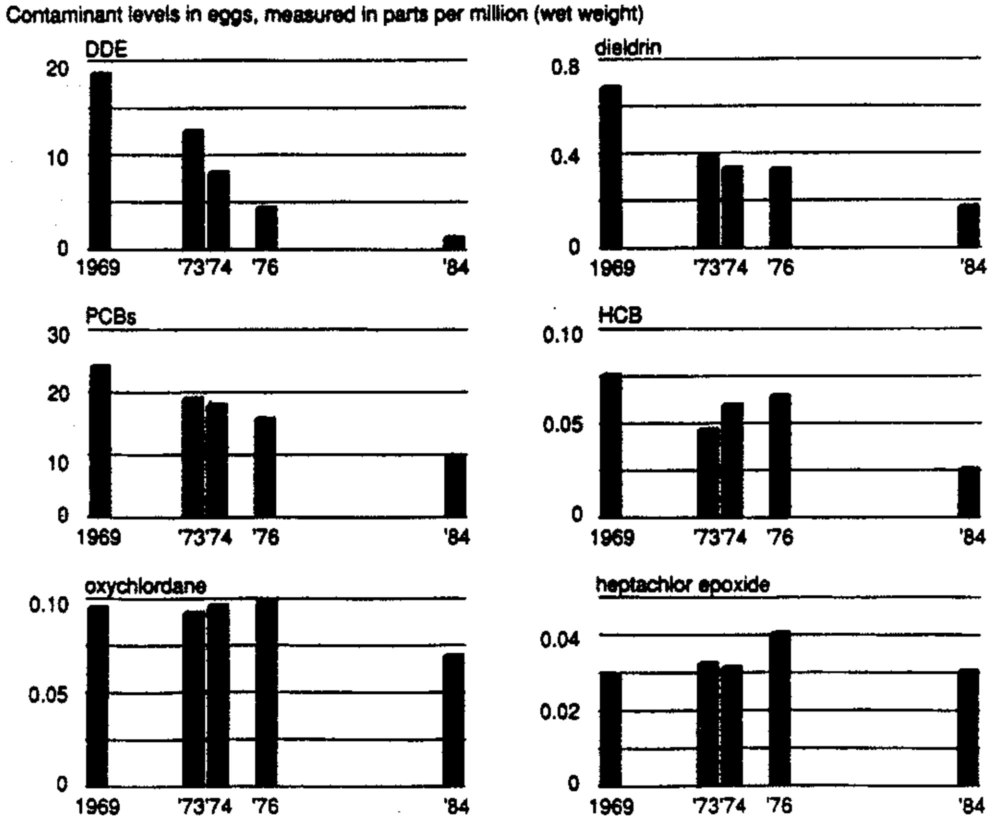


**Figure A7.7.** Temporal trends in organochlorine levels in gannet eggs from Quebec, Canada, from 1969–1984 (Chapdelaine *et al.*, 1987; Elliott *et al.*, 1988).




organochlorines in the aquatic environment, e.g., of PCBs in the Elbe River, is reflected immediately in the seabird egg concentrations (Figure A7.6). While levels of many organochlorines such as DDE, dieldrin, HCB, and PCBs have been shown to have decreased since the 1970s (e.g., Figures A7.5–A7.7), levels of some other organochlorines have shown no clear trend (Figure A7.7) and a few have increased (e.g., Becker *et al.*, 1998). Analyses of organochlorines in seabirds resident at high latitudes (Arctic and Antarctic) have provided evidence of the global transport of these contaminants, although concentrations in resident seabird species tend to be low, and less than in seabirds that breed in these regions but migrate to lower latitudes in winter (Lønne *et al.*, 1997; van den Brink, 1997).

While eggs tend to be sampled at a very clearly defined and consistent time of the year, avoiding problems of interseasonal variation, long-term trends in organochlorine concentrations in tissues of adult seabirds can be obscured where there are pronounced seasonal variations, and sampling across years is not limited to a short period (Joiris *et al.*, 1997).

It is worth emphasizing that the organochlorine concentrations in seabird eggs reflect local contamination in the vicinity of each breeding colony, even in seabird species that are transequatorial migrants such as the common tern *Sterna hirundo* (see Figure A7.4). After arrival in the breeding area, terns and other seabirds need large amounts of supplementary food presented by the males during the short prelaying period, enabling females to raise weight and produce eggs (common tern: up to 50 % weight increase during ten days, Wendeln and Becker, 1996). Thus, eggs provide a measure of contamination on a scale set by the foraging range of birds from their breeding colony. Exposure to these contaminants in the wintering area apparently has little or no influence on the amounts put into the egg. This may not be true for all seabird species, but it does appear to be a general pattern.

## 2.4 Metals and Organometallic Compounds

### 2.4.1 Mercury

Mercury is the heavy metal most likely to present a toxic hazard in marine foods. Particularly high concentrations occur in long-lived predatory and deep-sea fish. It is readily converted by bacteria from inorganic forms into methylmercury in low-oxygen environments (deep water or in anoxic sediments). Methylmercury is not only much more toxic to vertebrates, but is also lipid-soluble so it tends to biomagnify through food chains and is accumulated in lipid-rich tissues of vertebrates in a similar way to organochlorines (Tables A7.3–A7.5). As in the case of organochlorines, the within-seabird-sample variation is in the same order as the variation in fish, the main prey (Tables A7.3 and A7.4). Furthermore, while the assimilation efficiency of inorganic mercury from digested food is very low, the assimilation efficiency of methylmercury is around 95 %. Thus, most of the

**Table A7.3.** Mean concentrations ($\mu g\ g^{-1}$ wet weight) and coefficients of variation for mercury in marine organisms in Shetland. Median values are given for each group.

| Group | Species | Tissue | n | Mean ($\mu g\ g^{-1}$ wet weight) | CV | Ref. |
|---|---|---|---|---|---|---|
| Fish | sandeel | whole | 18 | 0.04 | 25 % | 1 |
| | cod | whole | 79 | 0.05 | 40 % | 1 |
| | whiting | whole | 20 | 0.07 | 29 % | 1 |
| | plaice | whole | 25 | 0.03 | 33 % | 1 |
| | | **Medians** | | **0.045** | **31 %** | |
| Seabirds | guillemot | chick down | 29 | 1.24 | 22 % | 2 |
| | kittiwake | chick down | 12 | 1.43 | 26 % | 2 |
| | Arctic tern | chick down | 24 | 2.03 | 32 % | 2 |
| | Arctic skua | chick down | 36 | 2.00 | 45 % | 2 |
| | great skua | chick down | 58 | 4.15 | 34 % | 2 |
| | | **Medians** | | **2.0** | **32 %** | |
| | kittiwake | chick feathers | 26 | 0.37 | 32 % | 2 |
| | Arctic tern | chick feathers | 15 | 0.69 | 20 % | 2 |
| | Arctic skua | chick feathers | 30 | 0.46 | 47 % | 2 |
| | great skua | chick feathers | 28 | 1.22 | 31 % | 2 |
| | | **Medians** | | **0.55** | **32 %** | |
| | fulmar | adult body feathers | 12 | 1.1 | 27 % | 1 |
| | kittiwake | adult body feathers | 14 | 2.4 | 25 % | 1 |
| | kittiwake | adult body feathers | 21 | 3.31 | 36 % | 2 |
| | razorbill | adult body feathers | 16 | 2.1 | 14 % | 1 |
| | guillemot | adult body feathers | 17 | 1.5 | 27 % | 1 |
| | guillemot | adult body feathers | 34 | 0.99 | 34 % | 2 |
| | puffin | adult body feathers | 10 | 5.2 | 52 % | 1 |
| | great skua | adult body feathers | 197 | 7.0 | 73 % | 1 |
| | great skua | adult body feathers | 54 | 6.34 | 41 % | 2 |
| | Arctic tern | adult body feathers | 23 | 0.86 | 27 % | 2 |
| | Arctic skua | adult body feathers | 28 | 2.52 | 88 % | 2 |
| | | **Medians** | | **2.4** | **34 %** | |

Ref. = Reference: 1 = Thompson *et al.* (1991); 2 = Stewart *et al.* (1997).

**Table A7.4.** Mean concentrations (µg $g^{-1}$ wet weight) and coefficients of variation for mercury in marine organisms in the Azores. Median values are given for each group.

| Group | Species | Tissue | n | Mean (µg $g^{-1}$ wet weight) | CV | Ref. |
|---|---|---|---|---|---|---|
| Fish | *Macroramphosus scolopax* | Whole | 42 | 0.16 | 34 % | 1 |
| | *Scomber japonicus* | Whole | 4 | 0.27 | 26 % | 1 |
| | *Capros aper* | Whole | 19 | 0.44 | 71 % | 1 |
| | *Trachurus picturatus* | Whole | 20 | 0.45 | 81 % | 1 |
| | *Maurolicus muelleri* | Whole | 11 | 1.03 | 22 % | 1 |
| | *Electrona rissoi* | Whole | 10 | 0.97 | 44 % | 1 |
| | *Myctophum punctatum* | Whole | 6 | 0.96 | 27 % | 1 |
| | *Ceratoscopelus maderensis* | Whole | 14 | 1.20 | 9 % | 1 |
| | | **Medians** | | **0.7** | **30 %** | |
| Seabirds | Bulwer's petrel | Egg | 16 | 1.60 | 41 % | 3 |
| | Cory's shearwater | Egg | 23 | 0.51 | 21 % | 3 |
| | Common tern | Egg | 20 | 0.32 | 50 % | 3 |
| | Roseate tern | Egg | 17 | 0.45 | 43 % | 3 |
| | | **Medians** | | **0.5** | **42 %** | |
| | Cory's shearwater | Chick feathers | 7 | 4.2 | 18 % | 2 |
| | Yellow-legged gull | Chick feathers | 5 | 4.0 | 38 % | 2 |
| | Common tern | Chick feathers | 42 | 1.6 | 41 % | 2 |
| | Roseate tern | Chick feathers | 13 | 1.2 | 47 % | 2 |
| | | **Medians** | | **2.5** | **40 %** | |
| | Bulwer's petrel | Adult body feathers | 24 | 22.4 | 22 % | 2 |
| | Cory's shearwater | Adult body feathers | 40 | 6.3 | 33 % | 2 |
| | Little shearwater | Adult body feathers | 4 | 2.4 | 48 % | 2 |
| | Madeiran storm petrel (June breeders) | Adult body feathers | 25 | 9.5 | 23 % | 2 |
| | Madeiran storm petrel (November breeders) | Adult body feathers | 27 | 16.0 | 27 % | 2 |
| | Common tern | Adult body feathers | 28 | 2.4 | 28 % | 2 |
| | Roseate tern | Adult body feathers | 21 | 2.2 | 36 % | 2 |
| | | **Medians** | | **6.3** | **28 %** | |

Ref. = Reference: 1 = Monteiro *et al.* (1996); 2 = Monteiro *et al.* (1995); 3 = Monteiro *et al.* (1998).

**Table A7.5.** Biomagnification factors between mercury in food and in seabird feathers for populations in the Azores (from Monteiro *et al.*, 1998).

| Species | Mercury in food ($\mu g\ g^{-1}$ dry weight) | Mercury in adult body feathers ($\mu g\ g^{-1}$ fresh weight) | Biomagnification factor (fresh weight to fresh weight) |
|---|---|---|---|
| Bulwer's petrel | 0.318 | 22.3 | 225 |
| Madeiran storm petrel (hot season) | 0.243 | 11.1 | 146 |
| Madeiran storm petrel (cool season) | 0.432 | 17.4 | 129 |
| Cory's shearwater | 0.131 | 5.4 | 132 |
| Little shearwater | 0.72 | 3.1 | 138 |
| Common tern | 0.54 | 2.1 | 125 |

mercury taken into the tissues of fish, marine mammals, and seabirds is methylmercury. However, at least some marine mammals and probably some seabirds can demethylate methylmercury in order to store it in a relatively non-toxic, and non-labile, inorganic form in the liver (Thompson and Furness, 1989b).

Mercury concentrations increase with age in fish and in marine mammals, but not in seabirds (Furness *et al.*, 1990; Furness, 1993). Seabirds lose their mercury into growing feathers. All mercury in feathers is methylmercury (Thompson and Furness, 1989a), even in seabirds where the bulk of the mercury in the liver is inorganic (Thompson and Furness, 1989b). Concentrations of mercury in feathers vary according to the moult pattern. Feathers renewed first in the major autumn moult have the highest concentrations, and concentrations decrease as the moult progresses and the body pool of mercury is depleted (Furness *et al.*, 1986; Braune, 1987; Braune and Gaskin, 1987). A sample of several small body feathers provides a good measure of the mercury level in an individual bird, and is the most appropriate way of using feathers from live birds or museum skins for mercury monitoring. Female seabirds may have slightly lower concentrations of mercury than those found in males because they put some mercury into the eggs (Becker, 1992), but differences between the sexes tend to be small (Lewis *et al.*, 1993). Both laboratory experiments and oral dosing of wild birds in the field with mercury have shown that mercury concentrations in the blood and internal organs are closely related to the ingested dose, and that concentrations in feathers are dependent on the mercury level in the blood during feather growth, which itself is a function of mercury level in the diet.

Mercury concentrations in seabirds can be related to mercury levels in their prey (Monteiro *et al.*, 1995, 1998; Figure A7.8) and, in particular, show that seabirds feeding on mesopelagic prey have much higher mercury burdens than seabirds feeding in other food chains (Tables A7.4 and A7.5). This reflects a trend for mercury levels to be higher in fish from deeper water (Monteiro *et al.*, 1996), presumably due to methylation of mercury in deep low-oxygen water, thereby permitting greater assimilation of mercury into biota.

Geographical variations in mercury contamination can be seen from sampling seabird feathers, chick down, or eggs from different colonies. For example, Renzoni *et al.* (1986) showed higher levels of mercury in Cory's shearwaters *Calonectris diomedea* in Mediterranean colonies than in Atlantic colonies. Becker *et al.* (1993) showed that feathers from tern and gull chicks indicated local patterns of mercury pollution of the German North Sea coast attributable to river inputs of mercury into the southern North Sea. Also eggs clearly indicate spatial variation (Figure A7.4) and temporal trends of mercury in the marine environment (Becker *et al.*, 1998).

Joiris *et al.* (1997) found a strong increase in the mercury levels in guillemots in the southern North Sea through the winter, as these birds spend the summer in areas with much lower mercury exposure (the northwestern North Sea) (Figure A7.9).

Body feathers from adult seabirds can be used to show long-term trends in mercury contamination, as museums hold material dating back to the 1850s. Inorganic mercury-contaminated study skins can be separated from the methylmercury put into feathers by the birds by making a simple biochemical fractionation, so that only the mercury of biological relevance is measured. Such studies have shown approximately 400 % increases in mercury in seabirds from the UK coast (Thompson *et al.*, 1992; Figure A7.10) and in the Azores region (Monteiro and Furness, 1997; Figure A7.11), but not in southern hemisphere seabirds (Thompson *et al.*, 1993b). These patterns match closely with predictions from modelling of atmospheric transport of mercury from industrial sources (Mason *et al.*, 1994; Fitzgerald, 1995), which predict a four-fold increase in mercury in northern hemisphere ecosystems but little increase in the southern hemisphere. On the southern North Sea coast, the pattern of mercury levels in herring gull *Larus argentatus* feathers from 1880–1990 showed about a 300 % increase during the Second World War and a second wave of

**Figure A7.8.** Relationship between mean mercury concentrations in breast feathers of seabirds from the Azores and in the food of these birds sampled during the breeding season at the colony. BB: *Bulweria bulweria*; CD: *Calonectris diomedea*; OC: *Oceanodroma castro*; H: June breeders; C: November breeders; PA: *Puffinus assimilis*; SH: *Sterna hirundo*. From Monteiro *et al.*, 1998.

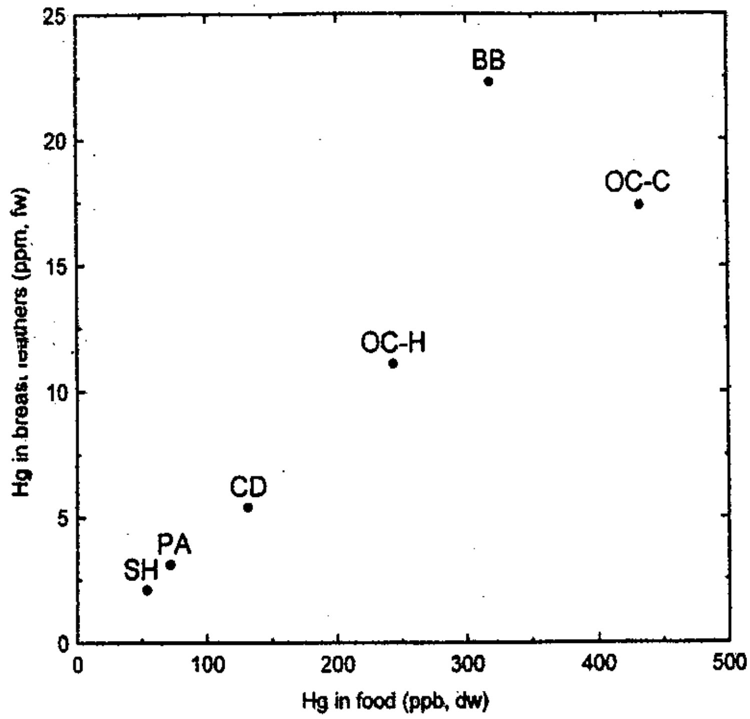



**Figure A7.9.** Mercury in common guillemot liver samples as a function of collection date ($\mu g\ g^{-1}$ dw; Joiris *et al.*, 1997).

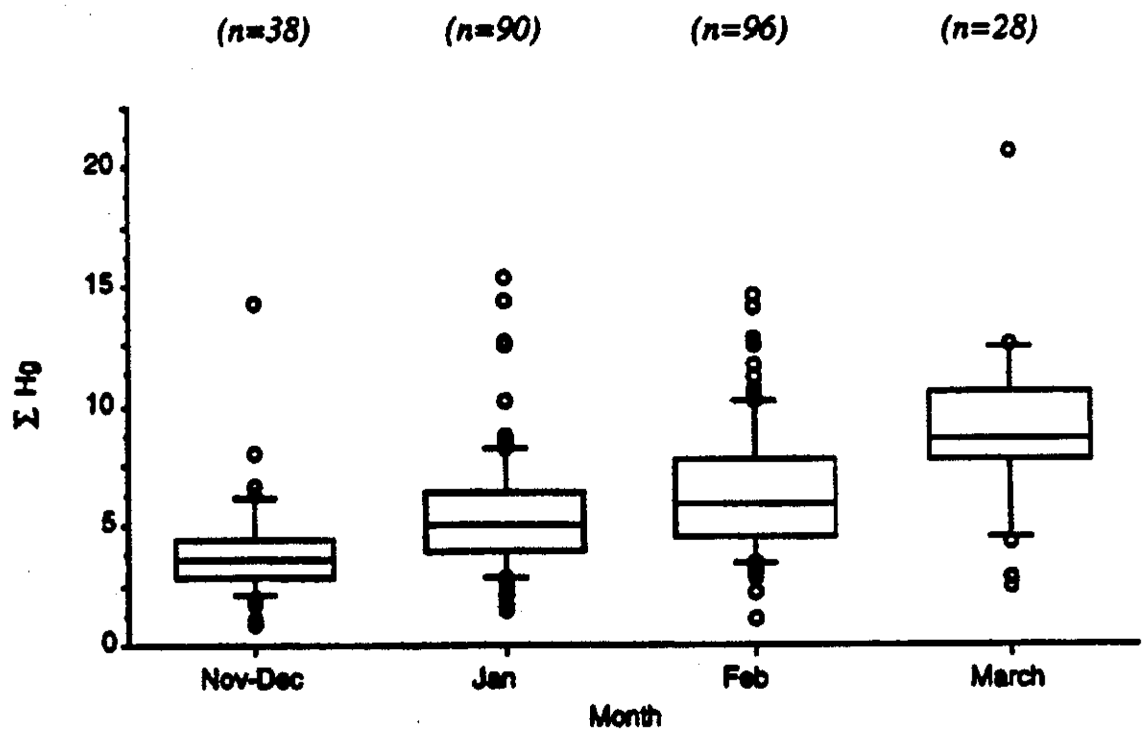


**Figure A7.10.** Mercury concentrations in body feathers of Atlantic puffins from southwest Britain and Ireland from 1850 to 1990 (from Thompson *et al.*, 1992).

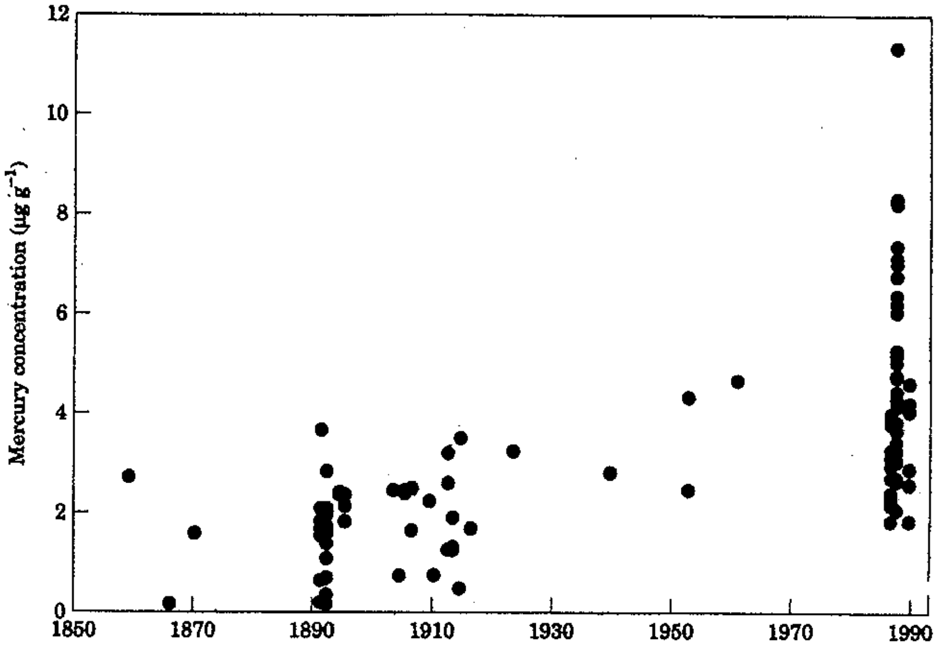
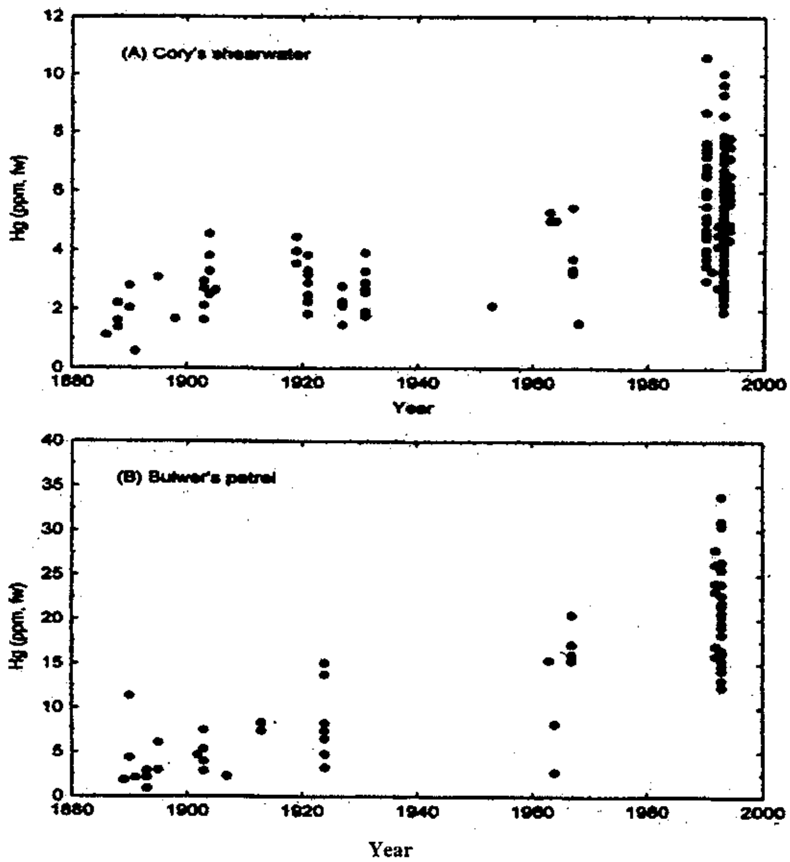



**Figure A7.11.** Mercury concentrations in body feathers of (A) Cory's shearwaters (which feed on epipelagic fish and squids) and (B) Bulwer's petrels (which feed on mesopelagic fish) in the Azores, from 1880 to 1995 (from Monteiro and Furness, 1997).



increase during the 1960s and 1970s owing to the industrial development in central Europe (Thompson *et al.*, 1993a). It appears that only seabird feathers permit retrospective monitoring of mercury contamination in marine food webs over the past 150 years (Swain *et al.*, 1992).

Whereas feathers reflect the bird's mercury body burden only during the time of feather growth, bird blood can be used indicating the present-day mercury contamination throughout the year (Kahle and Becker, in press).

### 2.4.2 Organotins

Although organotins (mainly tributyltin, TBT) have been mostly monitored in molluscs (Morcillo *et al.*, 1997; Harino *et al.*, 1998) and, with particular regard to imposex in whelks and developmental abnormalities in oysters, there has recently been increased interest in the pronounced bioaccumulation (especially in the liver) of organotins in marine mammals (Lee, 1991; Iwata *et al.*, 1997; Tanabe *et al.*, 1998) and in seabirds (Guruge *et al.*, 1997a; Kannan *et al.*, 1998). Factors resulting in high concentrations of organotins in particular species or populations of seabirds are not yet known and there is a need for further work on the patterns of accumulation of organotins by seabirds and the toxic implications of the accumulation. It is as yet unclear whether organotin contamination can be monitored by sampling seabirds. So far, organotin levels have been measured in liver and kidney tissues of seabirds and marine mammals, requiring sampling of dead animals or the killing of animals. Guruge *et al.* (1996, 1997b), however, show elevated TBT levels in feathers suggesting that birds excrete TBT, as they do mercury during moult, and that feathers could be used as an indicator of TBT contamination in wild birds.

### 2.4.3 *Other metals*

Cadmium is concentrated at high levels in the kidney of birds and mammals. Lead is concentrated particularly in the bones, but can also be measured in the blood. These elements enter eggs or feathers from the blood only in minute amounts, but cadmium and lead can be deposited onto feather surfaces from the atmosphere (Hahn, 1991); thus, using feathers to monitor amounts of these metals in the food chains of birds is confounded by problems of low concentrations and a high likelihood of external contamination. Nevertheless, feathers have been used to assess contamination by a wide range of elements (Burger, 1993). New techniques permitting the location of atoms within or on the surface of feathers may permit this practice to be developed with greater reliability. Several papers published in the 1970s and 1980s contain measurements of improbably high levels of metals in seabird feathers or eggs that must now be considered unreliable data.

Several papers provide details of concentrations of a range of elements in seabird tissues (e.g., Honda *et al.*, 1990; Elliott *et al.*, 1992; Wenzel and Gabrielsen, 1995; Kim *et al.*, 1996, 1998). This requires either the killing of birds to obtain samples or the use of chance sampling opportunities. Wenzel and Adelung (1996) examined the possible use of oiled birds as a means of sampling for heavy metal monitoring.

### 2.5 Radionuclides

Seabirds are probably not very useful in radionuclide monitoring because levels do not tend to increase up food chains and the assimilation efficiency of most radionuclides through the digestive system of seabirds is poor (Brisbin, 1993). Matishov *et al.* (1996) reported on caesium-137 in seabirds in the Barents Sea, but very few data on radionuclide levels in seabirds are available. One might anticipate that levels in mollusc-eating shorebirds and sea ducks could be elevated in areas such as the Cumbrian coast, but this does not seem to have been investigated.

## 3 ADVANTAGES OF SEABIRDS AS BIOMONITORS OF CONTAMINATION

The advantages that seabirds can provide as a tool for monitoring particular marine contaminants are considered below. Disadvantages are covered separately in Section 4, below.

### 3.1 *Taxonomy and Biology*

The phylogeny of seabirds has been the subject of very detailed research. There still remain some minor uncertainties, such as the numbers of taxa within certain groups. For example, molecular data suggest that some albatross species could be split into several closely related but genetically distinct species (Robertson and Gales, 1998). The Madeiran storm petrel *Oceanodroma castro* may consist of sibling species with seasonally distinct breeding (Monteiro and Furness, 1998). Nevertheless, such examples are aspects of detail, and it is unlikely that significant changes to the phylogeny of seabirds will arise as a result of further research. Thus, studies of contaminant levels can be based on a stable and well-described phylogeny. Furthermore, the huge amount of research on the biology of seabirds means that the migration patterns, seasonal distribution, feeding ecology, breeding biology, and physiology of seabirds are very well known. Of course, the amount of detail known about populations varies. There are some seabirds that have been little studied, whereas there are very large numbers of publications on the biology of some species such as herring gull, common guillemot *Uria aalge*, kittiwake *Rissa tridactyla*, and common tern. It is likely that widespread and well-studied species would also be most suitable as biomonitors of contaminants because a prerequisite for a biomonitor would be availability of satisfactory sample sizes and ease of sampling. The detailed knowledge of seabird ecology provides a good background for the interpretation of patterns of contaminant levels in seabirds, whereas for many other groups of marine animals, too little is known of the ecology of the organisms to permit such interpretation. Thus, for example, knowledge of the seasonal pattern of moult permits the selection of particular feathers to assess mercury contamination in different food webs in which the same individual bird is feeding at different times of the year (Thompson *et al.*, 1998a).

### 3.2 Tendency to Accumulate High Concentrations

Seabirds tend to feed at high trophic levels in marine food chains, so contaminants that accumulate up food chains will be well represented in seabirds. These include organochlorines and methylmercury, which are lipid-soluble contaminants with low solubility in water. Seabirds thus provide a potentially good biomonitor for lipid-soluble organic contaminants, since concentrations in seabirds are likely to be relatively high, and the careful selection of seabird species and sampling tissue should allow an appropriate spatial and temporal scale integration of the contaminant signal as well as giving an indication of the likely risk of toxic effects to animals high in the food chain (including man through harvesting of finfish and shellfish stocks). In contrast, many water-soluble contaminants with low lipid affinities, such as inorganic metals, show no trend of increased concentration with trophic level. In such cases, concentrations may be lower in seabirds than in some biota at low trophic levels. This is particularly true for radionuclides such as uranium and plutonium, where molluscs or algae provide a more appropriate biomonitor than do seabirds (Brisbin, 1993). Nevertheless, certain metals accumulate to high concentrations in particular avian tissues. For example, cadmium concentrations are particularly high in seabird kidneys (Stewart *et al.*, 1996).

As biomagnification factors not only increase with the food a species chooses, but also with the environmental burden from a contaminant, seabirds indicate inter-site or inter-year differences in contamination more distinctly than other animals (e.g., Figure A7.4).

### 3.3 Ease of Sampling

Almost all seabird species are colonial breeders, thus sampling large numbers of birds is often possible at selected colonies. Most seabirds breed at traditional sites every year, with the same adults usually nesting in the same territory each year, so that locations where seabird samples can be conveniently collected are highly predictable. When incubating, many adult seabirds are relatively easy to catch, and adults of some species are easy to catch while rearing chicks; but catching adults away from the nest and at times of the year when the birds are not breeding can be difficult. Eggs and chicks can be sampled at appropriate dates during the breeding season. Breeding tends to be consistent from year to year, so that optimal dates for sampling are predictable.

The behaviour of seabirds at the colony in response to human activity is highly variable from species to species. Birds of certain species panic and human disturbance can cause mortality of eggs or chicks, whereas other species are highly tolerant of disturbances. The choice of monitoring species and sites should take this into account. For example, cormorants tend to lose eggs or chicks when people enter their colony and they are not ideal as a choice of monitoring species for this reason. In contrast, kittiwakes tend to remain on the nest and egg or chick losses due to human disturbances are very rare. Responses can also vary between colonies. Adult gannets *Sula bassana* on the Bass Rock are easily caught at the nest with little disturbance, whereas gannets at Grassholm or St Kilda tend to panic when a human approaches. At the latter sites, gannets very rarely see humans at their colony, whereas at the Bass Rock human visitors are regular and numerous and the birds have learned to adapt to them.

Responses of particular species of seabirds to human intrusions are well known, as are the locations and accessibility of colonies, so it is very easily possible to plan, timetable, and cost a sampling programme.

By contrast, marine mammals are very difficult to sample, and most studies of contaminants in marine mammals have been based on small sample sizes of animals found stranded or entrapped in fishing gear (Addison *et al.*, 1984; Lee, 1991; Kannan *et al.*, 1994; Becker *et al.*, 1997; Fossi *et al.*, 1997; Iwata *et al.*, 1997; Krahn *et al.*, 1997; Moessner and Ballschmiter, 1997; Tanabe *et al.*, 1998). The concentrations of contaminants in marine mammals may be of interest because they give an indication of exposure that humans would experience from a marine diet, or from consumption of marine mammal meat (Weihe *et al.*, 1996), and they may reach levels that are toxic to marine mammals and so are of concern in terms of marine mammal conservation (Fossi *et al.*, 1997). However, in terms of monitoring marine ecosystems, sampling of marine mammals is difficult to achieve regularly and reliably. Seabirds are much more amenable to sampling in sufficient quantity.

### 3.4 Choice of Sampling Tissues

The ideal tissues to sample depend on the contaminant of interest, but the selection of several sampling tissues can often provide a much greater depth of information than taking a single monitoring tissue. For example, mercury concentrations vary among feathers of an individual seabird in a way that reflects the pattern of moult (Furness *et al.*, 1986; Braune, 1987; Braune and Gaskin, 1987), such that sampling feathers grown at different times of the year can indicate seasonal variations in mercury burdens of seabirds and, hence, seasonal patterns of mercury assimilation.

Adult seabirds tend to be long-lived and may range widely between breeding seasons, so that they may be exposed to contaminants far distant from the breeding colony. Migrations and diets may vary between individuals, between sexes, and between age classes, so that contaminant exposures may be highly variable between birds of differing status. With knowledge of these patterns, sampling can be planned to minimize the variance due to differences within populations. As a broad generalization, contaminant concentrations in chicks tend to be much less variable than those in adults. In part, this reflects the fact that contaminant burdens in chicks are largely derived from food fed to chicks during their growth, in addition to the amounts of contaminants received from their mothers prior to hatching (Becker *et al.*, 1993). Chick diets tend to be rich in energy-dense food, whereas adults may take a more varied diet and with greater variation in diet among individuals. Secondly, the food for the chicks is taken from a relatively small area over which parents forage during the chick-rearing period. In contrast, adults may range over much longer distances from the colony during the pre-laying and incubation periods, and may carry stored contaminants that they assimilated from food eaten during the non-breeding period when they were widely dispersed away from their colony. Sampling chicks can therefore provide indications of the level of contamination within the defined foraging range of the parents during chick-rearing and permit comparisons between colonies so that geographical variations in contaminant concentrations can be determined.

For some contaminants, samples of chick feathers or down (mercury) or blood (mercury, organochlorines, butyltins) provide satisfactory monitoring information. For others (e.g., cadmium), the concentrations in feathers or blood may be too low to quantify, since the contaminant is strongly bound in a particular tissue and is not free to circulate in the blood. Monitoring of such contaminants might require killing the chicks to obtain the necessary tissue (e.g., kidney for cadmium analysis).

Since killing chicks may be unacceptable, sampling eggs has its attractions, particularly as many species of seabirds can replace a clutch that is removed. Taking a single egg from a clutch of several may have very little effect on breeding success as the survival of the chicks from the remaining eggs may be increased due to the reduction in sibling competition. Sampling eggs takes little time and eggs are easy to handle and to store. Contaminant burdens in the eggs of a specific area and year tend to reflect the contaminant uptake of the female (healthy and reproductive members of the population) in the period shortly prior to egg laying (Becker, 1989; Dietrich *et al.*, 1997). Eggs can be used to measure contamination of the food web in the area around the colony over which food is gathered in the pre-laying period. Being restricted to the breeding season, the seasonal variability in levels of contaminants is reduced. Compared to tissues, the egg matrix has a consistent composition with high lipid contents that accumulate persistent compounds to high concentrations, simplifying chemical analyses.

### 3.5 Known Foraging Range

Although details of the foraging range of breeding seabirds are not known for every species, the information is available for many, and can be inferred for others from knowledge of closely related species or from other aspects of breeding biology (e.g., duration of the alternating periods spent on and off the egg during incubation). Foraging ranges may vary between different colonies or according to food abundance, but there is enough information known to permit sampling seabirds at breeding colonies selected to provide fairly accurate estimation of the geographical variation in contaminant concentrations. Foraging ranges of breeding seabirds vary from a few kilometres in the case of terns and shags, to thousands of kilometres, in the case of albatrosses and some shearwaters and petrels. Knowing the scale of foraging ranges of particular species may assist in selecting species that would provide the appropriate scale for a study. Terns, for example, can provide evidence of differences in contaminant levels resulting from local river discharges, whereas certain large albatrosses may provide data from the entire Southern Ocean.

### 3.6 Diet

Contaminant uptake will vary to some extent depending on the variability of the diet, both between individuals and across years. Selection of appropriate seabird species with narrow and consistent diets can avoid the noise that might otherwise be introduced by such variations in diet. For example, common guillemots and shags *Phalacrocorax aristotelis* have diets of fish that vary relatively little, whereas herring gulls are opportunists that may switch between highly differing diets. On this basis, herring gulls may be less suitable as biomonitors than common guillemots or shags. However, there are methods that can be used to investigate diet so that the effects of changes in diet can be assessed. These include both the conventional sampling of food regurgitates, fish observed to be carried into the colony, contents of pellets regurgitated by adults, samples offloaded from chicks by 'stomach-pumping', or indirect methods of diet assessment such as analysis of stable isotope ratios. Stable isotopes of carbon and of nitrogen have been widely used as indirect measures of diet, and especially of trophic status; they have recently been used in combination with analysis of contaminants in the same samples to aid the interpretation of differences in contaminant levels between samples (Hobson *et al.*, 1994; Jarman *et al.*, 1996, 1997; Atwell *et al.*, 1998). Since stable isotope analysis is based on the analysis of protein, it has the advantage that it can be used with feather material from study skins in museums so that even historical samples can be examined for dietary variation detectable by isotope analysis (Thompson and Furness, 1995; Thompson *et al.*, 1998a).

### 3.7 Historical Samples

Although there are few, and only rather recent, tissue banks that can provide materials for examining temporal trends in contaminant burdens in biota (Elliott, 1985; Schladot *et al.*, 1993; Becker *et al.*, 1997; Krahn *et al.*, 1997), museum materials can be of use as a means of examining long-term trends. Eggshells in museum collections provide clear evidence of the effects of DDT poisoning through eggshell thinning effects. Eggshells may also provide an opportunity to investigate trends in contaminant levels through chemical analysis of the shell or membrane composition, especially for some heavy metals (Burger, 1994; Burger and Gochfeld, 1996). Similarly, skeletal material might be used to examine trends in contamination, particularly for lead but possibly also for other contaminants. Feathers from study skins can be used to measure methylmercury contamination. Mercury concentrations in feathers reflect mercury levels in the blood at the time of feather growth and these in turn correlate with the amount of mercury in the diet (Monteiro and Furness, 1995; Monteiro *et al.*, 1998). All of the mercury assimilated by seabirds and subsequently excreted into growing feathers is in the form of methylmercury, so any later contamination of the feathers with inorganic mercury from dust or preservatives can be removed by a biochemical separation (Thompson and Furness, 1989a). As a result, it has been possible to quantify the increases in mercury contamination of marine food webs over the past 150 years by analysis of mercury concentrations in selected feathers from seabird skins (Thompson *et al.*, 1992, 1993a, 1993b, 1998b; Monteiro and Furness, 1997). Such an analysis is not possible for fish since fish are stored in preservative solutions that can affect concentrations of metals in the tissues. It might be possible to investigate long-term trends in concentrations of other heavy metals in seabird feathers (Burger, 1993), but this would require an analytical facility that can discriminate between metal incorporated into the feather structure from the bird's blood and metal that has been deposited onto the feather surface, either during the life of the bird or during storage.

### 3.8 Low Variance within a Population

Contaminant concentrations in samples of seabird chicks may be less variable than in other biota, such that the sample sizes required to detect a particular magnitude of increase would be less using seabirds than using other biota (Gilbertson *et al.*, 1987; Fryer and Nicholson, 1993). Similarly, selecting adult seabirds, or the chicks of seabirds with larger foraging ranges, may permit more cost-effective monitoring of long-term trends where spatial resolution of contaminant variation is not the objective. Where the aim is to examine spatial variation, seabirds are not as useful in providing the small-scale resolution that could be obtained using sedentary animals such as blue mussels; however, for large spatial scales (> 10 km), seabirds may integrate spatial variation that would be noise in an analysis based on sedentary animals and may provide a better means of assessing large spatial scale patterns.

### 3.9 High Public Interest

The fact that birds are of considerable public interest can be very helpful in a monitoring programme. Many amateur ornithologists, reserve wardens, and conservation staff are able to provide data or collect samples within a monitoring programme, and coordination of such work can be achieved through existing specialist groups such as the Seabird Group, Royal Society for the Protection of Birds (RSPB), scientific institutions or through wardens in nature reserves or national parks, and others. For example, in the UK, the Joint Nature Conservation Committee administers the seabird populations and productivity monitoring programme, with assistance from the Seabird Group, RSPB, English Nature, Scottish Natural Heritage, Countryside Council for Wales and others, to which over 100 people contribute data in a standardized format throughout the British Isles (Thompson *et al.*, 1997).

### 3.10 Resistance to Toxic Effects

As a broad generalization, seabirds appear to be more resistant to toxic effects of most contaminants than are mammals or terrestrial birds (Beyer *et al.*, 1996). High concentrations of mercury in apparently healthy breeding birds are found in many seabird species, well above levels that would cause toxic effects in terrestrial or freshwater birds (Thompson, 1996). PCBs can occur in concentrations in apparently healthy seabirds that would certainly have toxic effects at the same concentrations in mammals (Barron *et al.*, 1995; Guruge and Tanabe, 1997). On the other hand, during ontogeny seabirds are vulnerable to toxic chemicals like PCBs (e.g., Becker *et al.*, 1993). There is no evidence that TBT at moderately high levels has harmful effects in seabirds (Guruge *et al.*, 1997a; Kannan *et al.*, 1998).

The tendency for seabirds to be able to carry high concentrations of contaminants without displaying impaired reproduction or survival means that concentrations of contaminants in samples of seabirds should be a true reflection of exposure and not one that is biased by loss from the population of individuals carrying toxic doses. Choosing the egg as the sampling matrix avoids such problems, as eggs originate from the healthy, reproductive part of the population.

## 4 DRAWBACKS OF USING SEABIRDS AS BIOMONITORS OF CONTAMINATION

### 4.1 Complex Physiology

The complex physiology of vertebrates, in some respects, makes them less suitable than invertebrates as biomonitors. Seabirds regulate tissue concentrations of essential metals and partly regulate the concentrations of some non-essential metals. For example, the coefficient of variation for cadmium in seabirds is much less than for mercury. For monitoring purposes, seabirds would be of little or no use as monitors of iron, zinc or copper contamination, and may be less suitable than invertebrates as monitors of cadmium. Moult, seasonal variation in organ size, adaptation to season, fat deposition, anorexia, and other processes can affect contaminant concentrations in tissues, making changes between samples difficult to interpret as due to changes in pollution load rather than changes in physiology (van den Brink *et al.*, 1998).

Sex differences can occur, especially where females can excrete a contaminant into developing eggs, but such sex differences in contaminant burdens tend to be small (Furness *et al.*, 1990; Lewis *et al.*, 1993; Stewart *et al.*, 1994, 1997; Burger, 1995).

Eggs may be formed directly from recently assimilated food, or from materials drawn from stores within tissues. The relative importance of these two sources of material may vary between species, and often varies within a female through the laying sequence of egg production. As a result, since body stores and current diet may differ in contaminant content, egg mercury concentrations in terns and gulls decline with laying sequence (Becker, 1992). Levels of organochlorines also vary systematically with egg laying sequence (Mineau, 1982; Becker and Sperveslage, 1989).

### 4.2 Uncertain Provenance

Seabirds sampled at sea are unlikely to be all from a single local population but may include birds of differing status from a variety of breeding areas. Seabirds sampled at a colony may be more homogeneous, but their previous movements during the non-breeding season may have exposed them to various different sources of contamination and these may continue to be represented in the body burden through to the breeding season, possibly causing confusion.

### 4.3 Need to Avoid Killing Birds

In most countries, a licence is required to kill adult birds or chicks, to collect samples of eggs or to draw blood samples, or indeed to catch and handle live seabirds. However, such requirements should not hinder contaminant monitoring that is based on non-lethal sampling. In many countries in Europe, sampling by killing birds for contaminant analyses would now be considered to be unacceptable in most situations. The trend towards greater protection for wildlife is likely to continue. This could especially affect any programme of contaminant monitoring in countries where killing is currently an acceptable approach but may not be in the future, and any new programme of contaminant monitoring should be designed with such trends in public attitude and legislative control in mind.

### 4.4 Diet Switching and Specialization

Many seabirds will switch their diet according to the relative abundance of preferred prey, and such diet switching can affect contaminant burdens in seabirds, especially if the switch is between vertebrate and invertebrate prey types, or between prey at different trophic levels. Within populations, diet specialization can lead to increased variance of contaminant levels among individuals. Although seabird species can be selected as biomonitors on the basis of their having a stenophagous diet, even the most specialized seabirds may switch from one prey species to another if food availability changes enough. Such changes can be detected by monitoring the diet or by analysis of stable isotopes as indicators, but this adds to the cost and complexity of a monitoring programme.

### 4.5 Difficulties to Monitor Toxicity

Since seabirds tend to show a higher tolerance of many contaminants than do other animal groups, the opportunity to use breeding performance as a means of monitoring contaminant levels is of limited value. However, this possibility should not be discounted. Fox and Weseloh (1987) and Fox *et al.* (1991) suggested that breeding performance of gulls on the Great Lakes may be a useful indicator of contaminant exposure in this highly polluted region. They suggest the possibility that low gull breeding success might indicate toxic effects of a complex mixture of chemical contaminants that it would be extremely difficult and expensive to monitor by chemical analysis of gull samples or of other biota or physical samples. Thyen *et al.* (1998) proposed a programme for monitoring breeding success of coastal birds in the Wadden Sea, among other aims also to indicate contamination. However, seabird breeding failures can be caused by a wide variety of environmental factors, including food shortages, adverse weather, predators and human disturbances, so one should be cautious about the possibility of detecting a relatively weak signal due to contaminants from the considerable and often unpredictable noise caused by a wide range of other factors. One reproductive parameter which may indicate shell thinning or embryotoxicity by chemicals is hatching success, in case of its reduction after external causes have been excluded (Becker *et al.*, 1993).

It is possible that specific biomarkers of toxicity might be useful, either at the biochemical level (Peakall, 1992) or at the level of specific effects on reproduction. Sex ratio distortion as a result of feminization of genetically male embryos, for example, provides a fairly specific effect of oestrogenically active contaminants that is unlikely to be mimicked by other environmental influences. Similarly, teratogenic effects of particular contaminants may be evident in embryos or recently hatched chicks, while in uncontaminated populations, even those stressed by various natural environmental factors, such abnormalities are exceedingly rare.

## 5 CRITERIA FOR SELECTING SEABIRD BIOMONITORS

A good candidate for a seabird biomonitor of contaminants should have the following attributes:

a) accumulation of the contaminant to high concentrations;

b) resistance to toxic effects due to the contaminant (unless these are what is being monitored);

c) known, or preferably no, migratory habits;

d) a foraging range consistent with the spatial scale over which the contaminant is to be monitored;

e) a large population size with known breeding biology and ecology, and with large numbers of colonies throughout the area where contaminant monitoring is required;

f) be easy to collect without major disturbance to the breeding colony, and have easily identifiable life-stages if a particular category is to be sampled;

g) have known physiology;

h) have a narrowly defined and consistent diet;

i) feed predominantly or exclusively on prey in the food web under investigation.

Based implicitly or explicitly on criteria similar to these, the Institute of Terrestrial Ecology sampled the eggs of common guillemots and gannets to monitor organochlorine contamination of marine ecosystems around the United Kingdom. Gilbertson *et al.* (1987) proposed the monitoring of contaminants in the North Atlantic by the sampling of eggs of Atlantic puffin *Fratercula arctica* and/or common guillemot (pelagic fish food web), and Leach's storm petrel (plankton food web, west Atlantic). For the Wadden Sea, Becker (1989) and Becker *et al.* (1991) proposed common tern (fish) and oystercatcher (benthic animals) as monitoring species of the parameter 'contaminants in coastal bird eggs'. This parameter, of high priority within the Trilateral Monitoring and Assessment Program (TMAP), has been studied in the Wadden Sea every year since 1996 (Becker *et al.*, 1998). Choosing more than one seabird species feeding on different prey can indicate contamination of different parts of the food web.

In practice, most published studies reporting contaminant concentrations in seabirds, or even reporting spatial or temporal patterns of contaminant levels in seabirds, have been based on data apparently collected adventitiously rather than as a planned monitoring programme.

## 6 RECOMMENDATIONS FOR MONITORING CONTAMINANTS USING SEABIRDS

### 6.1 Oil

Ongoing programmes of monitoring the proportions of dead seabirds found on shorelines ('Beached Bird Surveys') should be encouraged as a cost-effective means (most are carried out by amateurs at no cost and are organized by NGOs) of determining long-term trends and geographical patterns of oil pollution at sea. Such monitoring is of greater interest to seabird conservation than to fisheries.

### 6.2 Plastic Particles

Given that this type of pollution appears to be increasing, there is a need for monitoring the amounts of plastic ingested by seabirds, especially petrels. Whether ingestion of plastic pellets by fish results in harm to the fish is unclear. The evidence from seabirds suggests that the plastic is directly ingested by seabirds and not obtained indirectly inside prey that they consume. Such monitoring is of greater interest to seabird conservation than to fisheries, but would help to increase public awareness and concern about plastic pollution of the seas.

### 6.3 Organochlorines

Sampling of seabird eggs as a means of monitoring local contamination by organochlorines (for advantages, see Section 2.3) should be developed into integrated marine contaminant monitoring programmes, with the selection of appropriate locally common and internationally widespread monitoring species. Table A7.7 lists seabird species suggested as monitors. In the Wadden Sea, the benthivorous oystercatcher *Haematopus ostralegus* and the common tern were selected as monitoring species within the regional Trilateral Monitoring and Assessment Program, which was implemented in 1996.

### 6.4 Mercury

Developed methods to monitor mercury contamination in marine food chains by sampling chick down or feathers from chicks or adults, or from blood or egg samples, should be applied in areas where there is concern about possible contamination of marine food chains by mercury. Sampling eggs from colonies located near rivers carrying mercury contamination can be used as a means of monitoring trends in riverine mercury loadings reaching the sea. Such monitoring provides useful evidence of the successful reduction in mercury contamination where technical measures have been implemented to reduce discharges. The same monitoring species used to measure organochlorine levels are proposed for integration in monitoring programmes for mercury (Table A7.7).

**Table A7.7.** Seabird species suggested as monitors of marine contamination by organochlorines and mercury in the Northeast Atlantic and adjacent seas. Information on population size and trends in Europe, clutch size, diets, feeding range as well as distribution is presented.

| | |
|---|---|
| **Fulmar** | 1 egg, not replaced if taken; 3 million pairs; populations increasing<br>wide-ranging pelagic: zooplankton, offal, discards, fish, squid<br>Norway, Iceland, Faroe, UK (all coasts), Ireland, France; North America, Greenland |
| **Gannet** | 1 egg; 230,000 pairs; populations increasing<br>wide-ranging: fish, sandeel, sprat, herring, mackerel, discards<br>Norway, Iceland, Faroe, UK (all coasts), Ireland, France; North America |
| **Shag** | 3–4 eggs, 86,000 pairs; populations mostly stable<br>coastal, short range: sandeel, sprat<br>Norway, Iceland, Faroe, UK (all coasts), Ireland, France, Mediterranean countries |
| **Kittiwake** | 2 eggs, 2–3 million pairs; populations mostly increasing or stable<br>wide ranging: small fish, zooplankton<br>Norway, Iceland, Faroe, UK (all coasts), Ireland, France, Helgoland, North America, Greenland |
| **Common tern** | 2–3 eggs, 208,000 pairs; some populations increasing, some stable, some decreasing<br>coastal: small fish<br>all European coasts (except Ireland and Faroe); North America |
| **Guillemot** | 1 egg, 2 million pairs; most populations increasing<br>inshore: fish, especially sandeel, sprat<br>Norway, Iceland, Faroe, UK (all coasts), Ireland, France, Sweden, Helgoland, North America |

### 6.5 Organotins

There is a need for research into organotin levels in seabirds to determine whether they may have toxic effects on seabirds, and whether seabirds may be used as a means of monitoring organotin contamination on larger scales.

### 6.6 Other metals

There is a need for research into the possible use of eggshells, egg contents or feathers for monitoring cadmium, lead, and other elemental concentrations to avoid the need to kill birds for liver, kidney or bone samples. In particular, if methods can be developed to measure elemental concentrations within feather keratin separately from contaminants on feather surfaces, this would permit retrospective monitoring of long-term trends in elemental contamination of marine food chains, which has been done successfully for mercury.

### 6.7 Radionuclides

Seabirds are not preferred organisms to monitor radionuclide contamination.

## 7 ACKNOWLEDGEMENT

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**Scientific names of bird species (in alphabetical order) mentioned in Annex 7.**

| **Common Name** | ***Scientific Name*** |
|---|---|
| Arctic skua | *Stercorarius parasiticus* |
| Arctic tern | *Sterna paradisaea* |
| Atlantic puffin | *Fratercula arctica* |
| Bulwer's petrel | *Bulweria bulweria* |
| Common guillemot | *Uria aalge* |
| Common tern | *Sterna hirundo* |
| Cory's shearwater | *Calonectris diomedea* |
| Fulmar | *Fulmarus glacialis* |
| Gannet | *Morus bassanus* |
| Great skua | *Catharacta skua* |
| Guillemot | *Uria aalge* |
| Herring gull | *Larus argentatus* |
| Kittiwake | *Rissa tridactyla* |
| Leach's storm petrel | *Oceanodroma leucorhoa* |
| Little shearwater | *Puffinus assimilis* |
| Madeiran storm petrel | *Oceanodroma castro* |
| Oystercatcher | *Haematopus ostralegus* |
| Puffin | *Fratercula arctica* |
| Razorbill | *Alca torda* |
| Roseate tern | *Sterna dougallii* |
| Shag | *Phalacrocorax aristotelis* |
| Yellow-legged gull | *Larus cachinnans* |

## ANNEX 8

## OVERVIEW OF INTERCALIBRATION/INTERCOMPARISON EXERCISES ON CHEMICAL ANALYSES COORDINATED BY ICES

*[Note: As ICES is no longer coordinating the conduct of intercalibration exercises, this Annex will not appear in future ACME reports.]*

### TRACE METALS IN BIOTA

**First ICES Intercalibration Exercise on Trace Metals in Biological Tissue**
(1/TM/BT) 1972

| | | |
|---|---|---|
| Coordinator | : | G. Topping, United Kingdom. |
| Sample | : | Fish flour prepared from commercial fish meal. |
| Metals analysed | : | Hg, Cu, Zn, Cd and Pb. |
| Participation | : | 8 laboratories from 7 countries around the North Sea. |

Results published in *Cooperative Research Report* No. 80 (1978).

**Second ICES Intercalibration Exercise on Trace Metals in Biological Tissue**
(2/TM/BT) 1973

| | | |
|---|---|---|
| Coordinator | : | G. Topping, United Kingdom. |
| Samples | : | Fish flour prepared from unskinned muscle of inshore cod and acidified solution of metals. |
| Metals analysed | : | Hg, Cu, Zn, Cd and Pb. |
| Participation | : | 15 laboratories in 11 countries around the North Sea and the Baltic Sea. |

Results published for Baltic Sea laboratories in *Cooperative Research Report* No. 63 (1977) and for North Sea laboratories in *Cooperative Research Report* No. 80 (1978).

**Third ICES Intercalibration Exercise on Trace Metals in Biological Tissue**
(3/TM/BT) 1975

| | | |
|---|---|---|
| Coordinator | : | G. Topping, United Kingdom. |
| Samples | : | (a) Fish flour prepared from skinned muscle of distant water cod, and<br>(b) individual reference standard solutions for each metal. |
| Metals analysed | : | Hg, Cu, Zn, Cd and Pb. |
| Participation | : | 29 laboratories in 17 ICES Member Countries. |

Results published for Baltic Sea laboratories in *Cooperative Research Report* No. 63 (1977) and for North Sea laboratories in *Cooperative Research Report* No. 80 (1978).

**Fourth ICES Intercalibration Exercise on Trace Metals in Biological Tissue**
(4/TM/BT) 1977

| | | |
|---|---|---|
| Coordinator | : | G. Topping, United Kingdom. |
| Samples | : | Same fish flour as in 3/TM/BT. |
| Metals analysed | : | Cd and Pb. |
| Participation | : | 12 of the laboratories which had participated in 3/TM/BT. |

Results published in *Cooperative Research Report* No. 108 (1981).

## Fifth ICES Intercalibration Exercise on Trace Metals in Biological Tissue
(5/TM/BT) 1978

| | | |
|---|---|---|
| Coordinator | : | G. Topping, United Kingdom. |
| Samples | : | (a) Fish flour prepared from skinned muscle of distant water cod, and |
| | | (b) the same fish flour extracted to produce a lower Hg concentration. |
| Metals analysed | : | Hg, Cu, Zn, Cd and Pb. |
| Participation | : | 41 laboratories, including those associated with the Joint Monitoring Programme, from all 18 ICES Member Countries plus several laboratories in Australia. |

Results published in *Cooperative Research Report* No. 108 (1981).

## Sixth ICES Intercalibration Exercise on Trace Metals in Biological Tissue
(6/TM/BT) 1979

| | | |
|---|---|---|
| Coordinator | : | G. Topping, United Kingdom. |
| Samples | : | (a) White meat of edible crab freeze-dried and ground into powder, |
| | | (b) commercial fish meal freeze-dried and ground into powder, and |
| | | (c) digestive gland of Canadian lobster treated and ground into powder. |
| Metals analysed | : | Hg, Cu, Zn, Cd and Pb. |
| Participation | : | 52 laboratories from 17 ICES Member Countries plus Australia. |

Results published in *Cooperative Research Report* No. 111 (1981).

## Seventh ICES Intercalibration Exercise on Trace Metals in Biological Tissue – Part 1
(7/TM/BT-1) 1983

| | | |
|---|---|---|
| Coordinators | : | S.S. Berman and V.J. Boyko, Canada. |
| Samples | : | (a) Lobster hepatopancreas homogenized, spray-dried and acetone extracted, |
| | | (b) scallop adductor muscle freeze-dried and ground, and |
| | | (c) plaice muscle freeze-dried and ground. |
| Metals analysed | : | Hg, Cu, Zn, Cd, As and Pb. |
| Participation | : | 51 laboratories from 17 ICES Member Countries. |

Results published in *Cooperative Research Report* No. 138 (1986).

## Seventh ICES Intercalibration Exercise on Trace Metals in Biological Tissue – Part 2
(7/TM/BT-2) 1985

| | | |
|---|---|---|
| Coordinators | : | S.S. Berman and V.J. Boyko, Canada. |
| Samples | : | (a) Cod liver, acetone-extracted and freeze dried, |
| | | (b) dogfish muscle, acetone-extracted and freeze dried, |
| | | (c) dogfish liver, acetone-extracted and freeze dried, |
| | | (d) whole dogfish, spray-dried, and |
| | | (e) *Mytilus edulis* soft material, freeze dried. |
| Metals analysed | : | Hg, Cu, Zn, Cd, As and Pb. |
| Participation | : | 49 laboratories from 16 ICES Member Countries. |

Results published in *ICES Cooperative Research Report* No. 189 (1992).

## TRACE METALS IN SEA WATER

### First ICES Intercalibration Exercise for Trace Metals in Sea Water
(1/TM/SW) 1976

| | | |
|---|---|---|
| Coordinator | : | P.G.W. Jones, United Kingdom. |
| Samples | : | Two standard solutions of metals. |
| Metals analysed | : | Hg, Pb, Ni, Co, Fe, Cr, Cu, Cd, Zn and Mn. |
| Participation | : | 41 laboratories from 14 ICES Member Countries. |

Results published in *Cooperative Research Report* No. 125 (1983).

### Second ICES Intercalibration Exercise for Trace Metals in Sea Water
(2/TM/SW) 1976

| | | |
|---|---|---|
| Coordinator | : | J. Olafsson, Iceland. |
| Samples | : | Two natural sea water samples and a mercury-spiked sea water sample, all acidified. |
| Metal analysed | : | Hg |
| Participation | : | 14 laboratories from 10 ICES Member Countries. |

Results published in *Cooperative Research Report* No. 125 (1983).

### Third ICES Intercalibration Exercise for Trace Metals in Sea Water
(3/TM/SW) 1977

| | | |
|---|---|---|
| Coordinator | : | P.G.W. Jones, United Kingdom. |
| Samples | : | Two frozen samples of filtered sea water, one from open North Sea waters and one from coastal waters. |
| Metals analysed | : | Co, Fe, Ni, Pb, Cd, Cr, Cu, Mn, and Zn. |
| Participation | : | 49 laboratories from 14 ICES Member Countries. |

Results published in *Cooperative Research Report* No. 125 (1983).

### Fourth ICES Intercalibration Exercise for Trace Metals in Sea Water
(4/TM/SW) 1978

| | | |
|---|---|---|
| Coordinators | : | J.M. Bewers, J. Dalziel, P.A. Yeats, and J.L. Barron, Canada. |
| Samples | : | Sets of six sea water samples consisting of four replicate sea water samples, one sample spiked with relevant metals and one dummy. Samples were frozen and acidified. |
| Metals analysed | : | Cd, Cu, Mn, Fe, Ni, Pb, and Zn. |
| Participation | : | 43 laboratories from 13 ICES Member Countries plus Monaco. |

Results published in *Cooperative Research Report* No. 105 (1981).

### ICES/JMG Intercalibration for Mercury in Sea Water
(ICES/JMG/1/HG/SW) 1979

| | | |
|---|---|---|
| Coordinator | : | J. Olafsson, Iceland. |
| Samples | : | (a) Two samples of natural sea water,<br>(b) sea water with a low Hg spike, and<br>(c) sea water with a high Hg spike. |
| Participation | : | 36 laboratories from all 13 Member Countries of the Oslo and Paris Commissions plus Canada, Japan, and the United States. |

Results published in *Cooperative Research Report* No. 110 (1981).

**ICES/JMG Intercalibration for Cadmium in Sea Water**
(ICES/JMG/1/CD/SW) 1979

| | | |
|---|---|---|
| Coordinator | : | Y. Thibaud, France. |
| Samples | : | (a) Natural sea water,<br>(b) sea water with a low Cd spike, and<br>(c) sea water with a high Cd spike. |
| Participation | : | 33 laboratories from all 13 Member Countries of the Oslo and Paris Commissions plus Canada and Monaco. |

Results published in *Cooperative Research Report* No. 110 (1981).

**Fifth ICES Intercalibration Exercise for Trace Metals in Sea Water**
(5/TM/SW:3 and 5/TM/SW:4) 1982–1983

| | | |
|---|---|---|
| Coordinators | : | J.M. Bewers, P.A. Yeats, S.S. Berman, D. Cossa, Canada; C Alzieu, P. Courau, France. |
| Samples | : | (a) Sea water samples, filtered and acidified, for analysis of metals except Hg, and<br>(b) sea water samples, natural and spiked, for analysis of Hg. In addition, 6 laboratories participated in an intercomparison of filtration procedures for coastal sea water samples. |
| Metals analysed | : | (a) Cd, Cu, Pb, Zn, Ni, Fe, Mn.<br>(b) Hg. |
| Participation | : | 59 laboratories from 15 ICES Member Countries plus Monaco. |

Results published in *Cooperative Research Report* No. 136 (1986).

**ICES Sixth Round Intercalibration for Trace Metals in Estuarine (Sea) Water**
(ICES/JMG/6/TM/SW) 1986

| | | |
|---|---|---|
| Coordinators | : | S. Berman, Canada; A. Jensen, Denmark; W. Cofino, The Netherlands. |
| Samples | : | (a) Sea water samples, filtered and acidified, for analysis of metals except Hg (salinity ca. 12),<br>(b) sea water samples, filtered and acidified, for analysis of metals except Hg (salinity ca. 28), and<br>(c) sea water samples, filtered and acidified, for analysis of Hg (salinity ca. 12). |
| Elements analysed | : | Cd, Cu, Hg, Zn. |
| Participation | : | 43 laboratories from 14 ICES Member Countries plus Italy. |

Results published in *Cooperative Research Report* No. 152 (1988).

**Seventh ICES Intercomparison Exercise on the Analysis of Trace Metals in Sea Water**
(7/TM/SW) 1996

| | | |
|---|---|---|
| Coordinators | : | B. Pedersen, Denmark; S. Berman, Canada. |
| Samples | : | (a) Natural brackish water samples (salinity 8),<br>(b) natural sea water samples (salinity 25). |
| Elements analysed | : | As, Cd, Co, Cr, Fe, Mn, Ni, Pb, Zn. |
| Participation | : | 39 laboratories from 12 ICES Member Countries plus three other countries. |

Results published in the *ICES Cooperative Research Report* No. 237 (2000).

## TRACE METALS IN MARINE SEDIMENTS

**First ICES Intercalibration Exercise for Trace Metals in Marine Sediments**
(1/TM/MS) 1984

| | | |
|---|---|---|
| Coordinator | : | D.H. Loring, Canada. |
| Samples | : | (a) Estuarine calcareous sandy mud sediment,<br>(b) harbour sediment, and<br>(c) Baltic mud sediment "MBSS" (from Baltic Sediment Intercalibration Exercise). |
| Metals analysed | : | Cd, Cr, Cu, Ni, Pb and Zn. |

| | | |
|---|---|---|
| Optional metals | : | Ti, Fe, Mn and Al. |
| Participation | : | 40 laboratories from 11 ICES Member Countries. |

Results published in *Cooperative Research Report* No. 143 (1987).

**Baltic Sediment Intercalibration Exercise–Step 1**
(Reference Samples ABSS and MBSS) 1983

| | | |
|---|---|---|
| Coordinators | : | L. Brügmann, German Democratic Republic, and L. Niemistö, Finland. |
| Samples | : | Two mud sediments ("ABSS" and "MBSS") from different locations, dried and homogenized. |
| Analytes | : | Cu, Pb, Zn, Cd, Mn, Fe, Cr, Ni, and organic C. |
| Optional | : | Hg, Co, Al, inorganic C, P and N. |
| Participation | : | 42 laboratories from 15 ICES Member Countries. |

Additional Exercise on Hg and Cd, 1985.

| | | |
|---|---|---|
| Coordinator | : | A. Jensen, Denmark. |
| Samples | : | Six samples, some of which were pre-treated. |
| Metals analysed | : | Hg and Cd. |
| Participation | : | 8 (Hg) and 10 (Cd) laboratories from 6 countries around the Baltic Sea. |

**Baltic Sediment Intercalibration Exercise–Step 2**
(Sliced Wet Cores) 1984

| | | |
|---|---|---|
| Coordinators | : | L. Brügmann, German Democratic Republic, L. Niemistö, Finland, and P. Pheiffer Madsen, Denmark. |
| Samples | : | 20 cm cores, sliced into 1-cm slices and deep frozen. |
| Main analytes | : | Cu, Cr, Zn, Pb, Mn, Cd, Fe, Ni, Al, Co, Hg, dry matter content, dating by Pb-210 technique. |
| Optional | : | Cs-137, organic C, N, P, clay minerals. |
| Participation | : | 11 laboratories from 6 countries around the Baltic Sea. |

Results for the entire exercise published in *Cooperative Research Report* No. 147 (1987).

## TRACE METALS IN SUSPENDED PARTICULATE MATTER

**First ICES Intercomparison Exercise for Trace Metals in Suspended Particulate Matter**
(1/TM/SM) 1984

| | | |
|---|---|---|
| Coordinators | : | P. Yeats and J.A. Dalziel, Canada. |
| Samples | : | Suspended particulate matter collected on pre-weighed 0.4 mm Nuclepore filters. |
| Analytes | : | Al, Fe, Mn, Zn, Cu, Pb, Ni, and Cd. |
| Participation | : | 8 selected laboratories from 7 countries. |

Results published in *Journal du Conseil International pour l'Exploration de la Mer*, 43: 272-278 (1987).

**Second ICES Intercomparison Exercise for Trace Metals in Suspended Particulate Matter–Phase 1**
(2/TM/SM-1) 1989

| | | |
|---|---|---|
| Coordinators | : | H. Hovind and J. Skei, Norway. |
| Samples | : | Standard reference materials from the National Research Council of Canada:<br>(a) PACS-1,<br>(b) MESS-1, and<br>(c) BCSS-1, from which participants should weigh out 1, 3, and 5 mg samples for analysis. |
| Analytes | : | Al, Cd, Cr, Cu, Fe, Mn, Ni, Pb, Zn. |
| Participation | : | 19 laboratories from 11 countries. |

Results published in *ICES Cooperative Research Report* No. 184 (1992).

**Second ICES Intercomparison Exercise for Trace Metals in Suspended Particulate Matter–Phase 2**
(2/TM/SM-2) 1993

| | | |
|---|---|---|
| Coordinator | : | C. Pohl, Germany. |
| Samples | : | Suspended particulate matter collected on pre-weighed 0.4 mm Nuclepore filters. |
| Analytes | : | Al, Cd, Co, Cu, Fe, Li, Mn, Ni, Pb, Zn. |
| Participation | : | 15 laboratories from 10 countries. |

Results published in *Accreditation and Quality Assurance*, 2: 2–10 (1997).

## ORGANOCHLORINES IN BIOLOGICAL TISSUE

**First ICES Intercalibration Exercise for Organochlorine Residues in Biological Tissue**
(1/OC/BT) 1972

| | | |
|---|---|---|
| Coordinator | : | A.V. Holden, United Kingdom. |
| Samples | : | (a) Natural fish oil, and |
| | | (b) same fish oil spiked with selected organochlorines. |
| Analytes | : | pp'-TDE, pp'-DDE, pp'-DDT, PCBs, dieldrin, γ-HCH |
| Participation | : | 9 laboratories from 7 ICES Member Countries. |

Results published in *Cooperative Research Report* No. 80 (1978).

**Second ICES Intercalibration Exercise for Organochlorine Residues in Biological Tissue**
(2/OC/BT) 1974

| | | |
|---|---|---|
| Coordinator | : | A.V. Holden, United Kingdom. |
| Samples | : | (a) unspiked maize oil, and |
| | | (b) same maize oil spiked with selected organochlorines. |
| Analytes | : | pp'-TDE, pp'-DDE, pp'-DDT, PCBs, dieldrin, γ-HCH |
| Participation | : | 30 laboratories from 13 ICES Member Countries. |

Results published in *Cooperative Research Report* No. 80 (1978) and, for Baltic laboratories, in *Cooperative Research Report* No. 63 (1977).

**Third ICES Intercalibration Exercise for Organochlorine Residues in Biological Tissue**
(3/OC/BT) 1978

| | | |
|---|---|---|
| Coordinator | : | A.V. Holden, United Kingdom. |
| Sample | : | Fish oil (capelin). |
| Analytes | : | pp'-TDE, pp'-DDE, pp'-DDT, PCBs, dieldrin, α-HCH, γ-HCH. |
| Participation | : | 30 laboratories from 16 ICES Member Countries. |

Results published in *Cooperative Research Report* No. 108 (1978).

**Fourth ICES Intercalibration Exercise for Organochlorine Residues in Biological Tissue**
(4/OC/BT) 1979

| | | |
|---|---|---|
| Coordinators | : | J.F. Uthe and C.J. Musial, Canada. |
| Samples | : | (a) Fish oil prepared from herring muscle tissue and |
| | | (b) same oil spiked with PCBs. |
| Analytes | : | PCBs |
| Participation | : | 23 laboratories from 12 ICES Member Countries. |

Results published in *Cooperative Research Report* No. 115 (1982).

**Fifth ICES Intercalibration Exercise for Organochlorine Residues in Biological Tissue**
(5/OC/BT) 1982

| | | |
|---|---|---|
| Coordinators | : | J.F. Uthe and C.J. Musial, Canada. |
| Samples | : | (a) Herring oil, and |
| | | (b) same oil spiked with individual chlorobiphenyls (CBs). |
| Analytes | : | Individual CBs. |
| Participation | : | 30 laboratories. |

Results published in *Cooperative Research Report* No. 136 (1986).

**Sixth ICES Intercalibration Exercise for Organochlorine Residues in Biological Tissue**
(6/OC/BT) 1983

| | | |
|---|---|---|
| Coordinators | : | L. Reutergårdh and K. Litzén, Sweden. |
| Samples | : | (a) Standard solution of 12 pure CBs, |
| | | (b) solution of an internal standard, and |
| | | (c) herring oil. |
| Analytes | : | Individual CBs. |
| Participation | : | 12 laboratories. |

Results published in *ICES Cooperative Research Report* No. 183 (1992).

**ICES/IOC/OSPARCOM Intercomparison Programme**
**on the Analysis of Chlorobiphenyls in Marine Media–Step 1**
(7/OC/BT-1 and 1/OC/MS-1) 1989

| | | |
|---|---|---|
| Coordinators | : | J. de Boer, The Netherlands (for ICES), J.C. Duinker, Federal Republic of Germany (for IOC), J.A. Calder, United States (for JMG). |
| Samples | : | (a) Standard solution of 10 CBs in iso-octane, |
| | | (b) solution of the 10 CBs in iso-octane at unknown concentration, |
| | | (c) internal standard: octachloronaphthalene in iso-octane, and |
| | | (d) blank: iso-octane. |
| Analytes | : | CB Nos. 28, 31, 52, 101, 105, 118, 138, 153, 180, 189. |
| Participation | : | 57 laboratories from 17 countries. |

Results published in *ICES Cooperative Research Report* No. 183 (1992).

**ICES/IOC/OSPARCOM Intercomparison Programme**
**on the Analysis of Chlorobiphenyls in Marine Media–Step 2**
(7/OC/BT-2 and 1/OC/MS-2) 1990

| | | |
|---|---|---|
| Coordinators | : | J. de Boer, The Netherlands (for ICES), J.C. Duinker, Federal Republic of Germany (for IOC), L. Reutergårdh, Sweden, and J.A. Calder, United States (for JMG). |
| Samples | : | (a) Standard solution (in iso-octane) of all CBs to be analysed; |
| | | (b) seal blubber extract in iso-octane; |
| | | (c) sediment extract in iso-octane; |
| | | (d) internal standard solution in iso-octane; and |
| | | (e) blank (iso-octane). |
| Analytes | : | CB Nos. 28, 31, 52, 101, 105, 118, 138, 153, 156, 180. |
| Participation | : | 58 laboratories from 16 countries. |

Results published in *ICES Cooperative Research Report* No. 207 (1995).

**ICES/IOC/OSPARCOM Intercomparison Programme on the Analysis of Chlorobiphenyls in Marine Media–Step 3a**
(7/OC/BT-3a and 1/OC/MS-3a) 1991

| | | |
|---|---|---|
| Coordinator | : | J. de Boer, The Netherlands. |
| Sample | : | Certified Reference Material CRM 349 cod liver oil (from the Community Bureau of Reference (BCR) of the European Community). |
| Analytes | : | CB Nos. 52, 153, 156. |
| Participation | : | 45 laboratories from 15 countries. |

Results published in *ICES Cooperative Research Report* No. 223 (1998).

**ICES/IOC/OSPARCOM Intercomparison Programme on the Analysis of Chlorobiphenyls in Marine Media–Step 3b**
(7/OC/BT-3b and 1/OC/MS-3b) 1992

| | | |
|---|---|---|
| Coordinators | : | J. de Boer and J. van der Meer, The Netherlands. |
| Samples | : | (a) A cleaned and an uncleaned sediment extract;<br>(b) a cleaned and an uncleaned seal blubber extract; and<br>(c) a standard solution. |
| Analytes | : | CB Nos. 28, 31, 52, 101, 105, 118, 138, 153, 156, 180. |
| Participation | : | 46 laboratories from 15 countries. |

Results published in *ICES Cooperative Research Report* No. 223 (1998).

**ICES/IOC/OSPARCOM Intercomparison Programme on the Analysis of Chlorobiphenyls in Marine Media–Step 4**
(7/OC/BT-4 and 1/OC/MS-4) 1993

| | | |
|---|---|---|
| Coordinators | : | J. de Boer and J. van der Meer, The Netherlands. |
| Samples | : | (a) Seal oil,<br>(b) sediment,<br>(c) Atlantic cod muscle,<br>(d) standard solution. |
| Analytes | : | CB Nos. 28, 31, 52, 101, 105, 118, 138, 153, 156, 180. |
| Participation | : | 43 laboratories from 15 countries. |

Results published in *ICES Cooperative Research Report* No. 223 (1998).

## HYDROCARBONS IN MARINE SAMPLES

**First ICES Intercomparison Exercise on Petroleum Hydrocarbons in Marine Samples**
(1/HC/BT and 1/HC/MS) 1980

| | | |
|---|---|---|
| Coordinators | : | R.J. Law and J.E. Portmann, United Kingdom. |
| Samples | : | (a) Crude oil standard,<br>(b) aliphatic fraction of crude oil standard,<br>(c) marine sediment, and<br>(d) mussel homogenate. |
| Analytes | : | Total hydrocarbons, aliphatic hydrocarbons ($nC_7$–$nC_{33}$), and several aromatic hydrocarbons. |
| Participation | : | 36 laboratories from 12 ICES Member Countries and Bermuda. |

Results published in *Cooperative Research Report* No. 117 (1982).

**ICES/IOC Intercomparison Exercise on Petroleum Hydrocarbons in Biological Tissues**
(2/HC/BT) 1984

| | | |
|---|---|---|
| Coordinators | : | J.W. Farrington, A.C. Davis, J.B. Livramento, C.H. Clifford, N.M. Frew, A. Knap, United States. |
| Samples | : | (a) Three samples of frozen, freeze-dried mussel homogenate,<br>(b) reagent-grade chrysene,<br>(c) methylene chloride solution of n-alkanes,<br>(d) methylene chloride solution of aromatic hydrocarbons, and<br>(e) Arabian Light Crude Oil standard. |
| Analytes | : | Aliphatic hydrocarbons ($nC_{15}$–$nC_{32}$) and selected aromatic hydrocarbons. |
| Participation | : | 38 laboratories from 13 ICES Member Countries and 12 laboratories from 11 IOC Member Countries (most, if not all, ICES Member Countries are also members of IOC). |

Results published in *Cooperative Research Report* No. 141 (1986).

**Third ICES Intercomparison Exercise on Polycyclic Aromatic Hydrocarbons in Biological Tissue**
(3/HC/BT) 1984

| | | |
|---|---|---|
| Coordinators | : | J.F. Uthe, C.J. Musial, and G.R. Sirota, Canada. |
| Samples | : | (a) Acetone powder of lobster digestive gland, and<br>(b) the oil extracted during the preparation of this powder. |
| Analytes | : | 21 selected polycyclic aromatic hydrocarbons. |
| Participation | : | 11 laboratories from 7 ICES Member Countries. |

Results published in *Cooperative Research Report* No. 141 (1986).

**Fourth ICES Intercomparison Exercise on Polycyclic Aromatic Hydrocarbons in Marine Media–Stage 1**
(4/HC/BT and 2/HC/MS) 1988–1990

| | | |
|---|---|---|
| Coordinator | : | R.J. Law, United Kingdom. |
| Samples | : | Solutions of 10 PAHs in acetonitrile (for HPLC analysis), or solutions of 10 PAHs in hexane (for GC analysis). |
| Analytes | : | Phenanthrene, fluoranthene, pyrene, benz[*a*]anthracene, chrysene, benzo[*e*]pyrene benzo[*a*]pyrene, benzo[*b*]fluoranthene, benzo[*ghi*]perylene, and indeno[123-*cd*]pyrene. |
| Participation | : | 17 laboratories from 9 countries. |

Report on results published in *ICES Cooperative Research Report* No. 207 (1995).

## NUTRIENTS IN SEA WATER

**Fourth ICES Intercomparison Exercise for Nutrients in Sea Water**
(4/NU/SW) 1989

| | | |
|---|---|---|
| Coordinators | : | D. Kirkwood, United Kingdom, A. Aminot, France, and M. Perttilä, Finland. |
| Samples | : | (a) Natural oceanic water, with no preservatives or pre-treatment,<br>(b) natural shelf sea water, filtered, bottled in glass and autoclaved, and<br>(c) sea water depleted in nitrate and phosphate, then filtered and bottled (blanks for nitrate and phosphate). |
| Analytes | : | Nitrate + nitrite, phosphate, silicate, nitrite, ammonia, total nitrogen and total phosphorus. |
| Participation | : | 68 laboratories from all 18 ICES Member Countries. |

Report on results published in *ICES Cooperative Research Report* No. 174 (1991).

**Fifth ICES Intercomparison Exercise for Nutrients in Sea Water**
(5/NU/SW) 1993

| | | |
|---|---|---|
| Coordinators | : | D. Kirkwood, United Kingdom, and A. Aminot, France. |
| Samples | : | Six samples of sea water (three for nitrate + nitrite determinations and three for ammonium and phosphate determinations). |
| Analytes | : | Nitrate + nitrite, ammonium, phosphate. |
| Participation | : | 132 laboratories from 31 countries. |

Report on results published in *ICES Cooperative Research Report* No. 213 (1995).

# ANNEX 9

## ACME/ACMP ADVICE BY TOPIC FOR THE YEARS 1988–1999

Numbers in the table refer to sections of the present report and of the ACMP or ACME reports from 1988 to 1999, in reverse chronological order.
*Signifies major advice on that topic.

| Topic | Sub-topic | 1999 | 1998 | 1997 | 1996 | 1995 | 1994 | 1993 | 1992 | 1991 | 1990 | 1989 | 1988 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Monitoring | Strategy | | | | | 5.1 | *4; *Ann. 1 | 5 | 5.1 | | | | *4 |
| | Programme evaluation | | | | | | 4.2 | | | | | | |
| | Statistical methods for design | 5.6 | | | | | | | | | | | |
| | Benthos | | | | 6.1.2; 11.1; *Ann. 8 | | | | 8; *Ann. 6 | 8.1 | 9 | *Ann. 1 | *7.1 |
| | NSTF/MMP | | | | | | | 5.2 | | | 9.3 | 4 | |
| | Sediments/guidelines | | 4.6; *Ann. 2 | 4.5; *Ann. 1 | 5.5; *Ann. 4 | | 5.5 | 6.1; *Ann. 1 | | | | *14 | |
| | Sediment data normalization | 5.5 | | 4.5.2 | 5.5.1 | | 5.5 | | | | | *14.1 | 14.1 |
| | Sediment sensitivity, variance factors | | | | | 5.6 | | | | | | | 14.6; Ann. 2 |
| | Metals/sediments | | | | 9.5 | 5.6 | 5.5 | | | | | | 12.6 |
| | Matrix tables | | | | | | | | | | | | |
| | • general (JMP) | | | | | | | | | | *Ann. 1 | *6.1 | 4 |
| | • organic | | | | | | | | | | 6.1 | | |
| | • NSTF | | | | | | | | | | 6.1 | | |
| | Substances that can be monitored | | | | | | | | | | | | |
| | • organic | 5.4 | 4.5 | | 5.4 | 6.6 | 6.8 | | | | | | |
| | • inorganic | 5.4 | 4.5 | 4.2 | | | | | | | | | |
| | Use of seaweeds | | | | | | 5.1 | | | | 6.8 | 6.3 | |
| | Use of seabird eggs | 13.2; Ann. 7 | 4.7.5 | | | | | | | | | | |
| | Spatial monitoring | | | *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 | 5.2 | 4.4 | | 5.1.3 | | 5.4 | | | | | | |
| | Effects of nutrient enrichment | 12.1 | | | 9.1 | 5.8 | | | | | | | |
| | Monitoring PAHs | | 4.2; *Ann. 1 | 4.4.1; 4.5; *Ann. 1 | | | | | | | | | |

| Topic | Sub-topic | 1999 | 1998 | 1997 | 1996 | 1995 | 1994 | 1993 | 1992 | 1991 | 1990 | 1989 | 1988 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Assessment | Statistical methods | | 4.7.1 | | | | | | | | | | |
| | Combined effects of contaminants | 5.1.5 | 11.2 | | | | | | | | | | |
| Temporal trend monitoring | Strategy/objectives | | | | *4; Ann. 1 | | *4; *Ann. 1 | | | | | | |
| | Guidelines | | | | 4.4 | | 4;5.2 | | | | | | 5.2 |
| | Data analysis | | 4.7.4 | 15.2 | | | 5.2 | 6.2 | 5.2 | | | | |
| | Nutrients | | | | 5.7 | | | 6.3 | | | | | 11.1 |
| | Fish/JMP/JAMP | 5.6.4 | | | 5.6 | | | | | 5.1 | 6.2 | | |
| | Fish/CMP | | | | 5.6 | | | | | 5.1 | 6.5 | *6.4 | 5.1 |
| | Biota/BMP | | | | | 7.3 | | | | | | | |
| | Biological effects | | 4.1 | | | | | | | | | | |
| | Mussels | | | | | | | | | 5.1 | *Ann. 3 | | |
| | Pooling | | | | | | | | | | 6.7; Ann. 5 | 6.4.3 | |
| | Precision | | | | | | 5.2 | | *Ann. 1 | | | 6.4.4 | |
| | Sediment storage | | | | | | | | | | | | 14.2 |
| | Sediments | | | | 4.3; 5.5.3 | | | | | | | | 5.3; Ann. 2 |
| | Statistical requirements | | 4.7 | | | 4.3 | | | | | | | |
| Integration of biological/ chemical measurements | Sediment quality | | | | 5.2.2 | 4.2; Ann. 2 | 5.4 | 6.4; *Ann. 2 | | | | | |
| Biological effects monitoring | Monitoring strategy | | | | *5.3 | 4.1; *Ann. 1 | | | | | | | |
| | Concepts | | | | | | | | | | | *7.1 | |
| | Statistical design | | 4.1* | 4.31 | | | | | | | | | |
| | Methods | 5.1.1 | | 4.3.2 | 5.3.2 | Ann. 1 | | | 6.2 | 7.2 | | | |
| | Molecular techniques | | | | | *5.2 | | | | | | | |
| | Pathology | | | 4.3.3 | 5.3.3 | 8.4 | 9.4 | | | | | | |
| | Workshop results | | | | 5.3.2 | | | | 6.1 | 7.1 | 8.1 | 7.2 | 6 |
| | Source of variability | 5.1.2 | | | | | | | | | | | |
| | Data analysis<br>• general | | | | | | | | 6.3 | | | | |
| | • EROD | | | | | | | | *Ann. 2 | | | | |
| | • oyster bioassay | | | | | | | | *Ann. 2 | | | | |

| Topic | Sub-topic | 1999 | 1998 | 1997 | 1996 | 1995 | 1994 | 1993 | 1992 | 1991 | 1990 | 1989 | 1988 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Baseline studies | ICES Baseline TM/SW | | | | | | | | | 6 | *7 | 6.5 | 13 |
| | Contaminants in • Baltic sediments | 5.2 | 4.3 | 6.1 | 7.1 | 7.1 | 7.1 | 8.1 | 13.2 | 14.1 | 15.1 | | |
| | • North Sea sediments | | | | | | | | 13.1 | | | | |
| | HCH in sea water | | | | | | | | 14 | | | | |
| Regional assessments | Guidelines | | | | | | | | | | 5 | | *20.1 |
| | Preparation plans | | | | | | | | | | | 5 | |
| | North Sea QSR | | | | | | | 4.1 | 4 | 4 | 5 | | |
| | Baltic Sea | | | | 7.2 | 7.2 | 7.3 | | | | | 5 | |
| | Baltic fish | | | | 7.3 | 7.2 | 7.3 | | | | 17.2 | 17.3 | |
| | Canadian waters | | | | | | | | | 16 | | | |
| | Nutrient trends–North Atlantic | | | | | | | | | | 13 | 12 | |
| Quality assurance | Philosophy | | | | | | | | | 13.6 | | | |
| | Reference materials | | 5.6 | 4.2 | | | *6.9 | 7.11 | | | | 13.1 | 12.8 |
| | Oxygen in sea water | | | *Ann. 3 | | | | | | | 14.5 | 13.6 | |
| | Nutrients | | | 5.7 | | | | | | | | | |
| | Quality/comparability • organic contaminants | 5.4 | 4.5 | | | *6.6 | *6.8 | | | | | | |
| | Hydrocarbons | | | | | | | | | | | 13.7 | |
| | Lipids | | | | | 6.4 | 6.5 | | | | | | |
| | NSTF | | | | | | | | | | 14.7 | | |
| | Biological effects techniques | 7.4 | 5.4 | 5.3 | *6.2; *Ann. 5 | 6.2 | | 7.1 | | 7.3 | | | |
| | Sediment quality criteria | | | | | | | | | | 15.2 | 22.2 | |
| | QA of sampling | 7.6 | 5.7 | 5.10 | | | | | *12.8 | | | | |
| | QA info. in data bank | | | 16.1.1 | | | 6.10 | | | | | | |
| | Chemical measurements–Baltic Sea | 7.5 | 5.5 | 5.4 | 6.3 | 6.3 | 6.2 | 7.4 | | | | | |
| | Biological measurements | 7.1; 7.2 | 5.1; 5.2 | 5.1; 5.2 | 6.1 | 6.1 | 6.1 | 7.3 | | | | | |
| | Fish disease monitoring | | 8.2 | 5.3.2 | *Ann. 6 | | | | | | | | |

| Topic | Sub-topic | 1999 | 1998 | 1997 | 1996 | 1995 | 1994 | 1993 | 1992 | 1991 | 1990 | 1989 | 1988 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Intercomparison exercises | Status | Ann. 8 | | Ann. 10 | Ann. 10 | Ann. 7 | Ann. 6 | Ann. 5 | Ann. 8 | Ann. 3 | Ann. 9 | Ann. 2 | Ann. 3 |
| | Nutrients/sea water | | | | 6.4 | 6.5 | 6.6 | 7.8 | 12.4 | | 14.1 | 13.4 | 11.3 |
| | Hydrocarbons in | | | | | | | | | | | | |
| | • biota | | | | | 6.7 | | | | | | 13.7 | |
| | • sediments | | | | | | | | | | | 13.7 | |
| | • sea water | | | | | | | | | | | 13.7 | |
| | PAHs/standards | | | | | | | | 12.2 | 13.1 | 14.2 | 13.2 | 12.1 |
| | PCBs/CBs in biota | | | | | 6.6; 6.7 | 6.3;6.4 | 7.5 | 12.1 | 13.2 | | | 12.3 |
| | Organochlorines in biota | | | | | 6.6; 6.7 | | | | | | | |
| | CBs/standards | | | | | | | | | | 14.3 | 13.3 | 12.2 |
| | CBs in sediments | | | | | 6.6 | 6.3 | 7.5 | | 13.2 | | | |
| | Metals in | | | | | | | | | | | | |
| | • sea water | | | 5.5 | 6.5 | | | | | | | | |
| | • sediments | | | | | | | | | | | | 12.5 |
| | • SPM | | | | | | 6.7 | 7.9 | 12.3 | 13.3 | 14.4 | 13.5 | 14.4; Ann. 1 |
| | Dissolved oxygen in sea water | | | | | | | | 12.5 | 13.4 | *14.5 | | |
| | Methyl Hg in biological tissue | | | | | | | | | | | | 12.4 |
| | Primary production | | | | | | | | | | | | 10.2 |
| | Oyster embryo bioassay | | | | | | | | Ann. 4 | | | | |
| | EROD | | | | | | | | 12.6; *Ann. 3 | | | | |
| Methods | Nutrients in sea water | | | | | | | | | | | 13.5 | *Ann. 4 |
| | DO in sea water | | | *Ann. 3 | | | | | | | | | |
| | Sediment normalization | | | | | | 5.5 | | | 14.2; 14.3 | | | |
| | Organic carbon measurements | | | 4.6; Ann. 2 | | | | | | | | | |
| Algal blooms | Primary production methods | | | | | | | 6.5 | 11 | 11.1 | 12.1 | | 10.2 |
| | Initiating factors | | | | | | | | | *11.3 | 12.2 | | |
| | Dynamics | | 10.2 | 9.2 | | | 8 | 10 | | | | | |
| | Exceptional blooms | 12.2; Ann. 2 | Ann. 3 | Ann. 8 | | | | | | 11.2 | | | |
| | Phycotoxins/ measurements | | | | | | | | | 11.4 | 12.3 | | 10.1 |
| | *C. polylepis* bloom | | | | | | | | | | | *11.1 | 10.3 |

| Topic | Sub-topic | 1999 | 1998 | 1997 | 1996 | 1995 | 1994 | 1993 | 1992 | 1991 | 1990 | 1989 | 1988 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Fish diseases and related issues | Relation to pollution | | 8.2 | 8.3 | 5.3.3 | 8.4 | 9.4 | | | 9.1 | | 9.3 | |
| | Survey methods | | | 7.2 | | | | | | 9.2 | | | 8.2 |
| | Diseases in wild fish | 10.1 | | | | | | | | | | | |
| | Baltic fish | | | 7.2 | | 8.1; 8.2; 8.3 | 9.3 | | | | | | |
| | Survey results | | | | | | | | | | | 9.1 | 8.1 |
| | Data analysis | 10.2; Ann. 5 | 8.1; *Ann. 8 | 7.1 | 8.2 | | 9.5 | 9.4 | 7 | | | | |
| | M74 in Baltic salmon | 10.3 | 8.3 | 6.2 | 7.4 | 7.4 | 9.1 | | | | | | |
| Mariculture | Interactions | 15.2 | | | 15.1 | 14 | 13 | | 9.1 | | *11 | 10 | 9 |
| | Escape of fish—effects | | 14.2 | 14.1 | | | | | | | | | |
| | Nutrient inputs/Baltic | | | | | | | | *9.2 | | 11.1 | 10.4 | |
| | Use of chemicals | 15.2 | | 14.2 | | | | | | | Ann. 6 | | |
| Introductions and Transfers | Code of Practice | | | | 14.2 | 13.1 | 14.1 | 12.1 | | | | | |
| | Accidental transfers | 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 | | | 13.1 | 14.1 | | | | | | | | |
| | Baltic Sea | 11.1 | 9.3 | | 14.3 | | | | | | | | |
| Marine mammals | Contaminants/effects | 14.1 | 12.2; *Ann. 10 | 11.4 | 5.4.2; 13.3; 13.4 | | | | | | | | |
| | Seal epidemic 1988 | | | | | | | | | | *18 | *18.1 | |
| | Baltic marine mammal stocks | 14.2 | | 11.1 | 13.1 | | 10.2 | | *18 | | | | |
| | Populations/N. Atlantic | | | | | | 10.1 | 11.1 | *18 | 18 | | | |
| | Pathogens | | | | | | | 11.2; Ann. 3 | | | | | |
| | Impact of fisheries | | 12.1; *Ann. 9 | 11.2; 11.3 | 13.2 | | | | | | | | |
| Seabirds | Diet, food consumption | 13.1; *Ann. 6 | | | | | | | | | | | |
| | Use in contaminant monitoring | 13.2; *Ann. 7 | | | | 5.3 | | | | | | | |
| Overviews | Mercury | | 7.1; *Ann. 4 | | | | | | | | | *19.1 | |
| | Hormone disruptors | | 7.4; Ann. 6 | | Ann. 2 | | | | | | | | |
| | HCB | | | | | | | | | | *20.1 | | |
| | Lindane (γ-HCH) | | | | | | | | | | *20.1 | | |

| Topic | Sub-topic | 1999 | 1998 | 1997 | 1996 | 1995 | 1994 | 1993 | 1992 | 1991 | 1990 | 1989 | 1988 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Overviews (cont.) | Tris(4-chlorophenyl) methanol/methane | | | | 10; *Ann. 7 | | | | | | | | |
| | Benzene/ alkylated benzenes | | | | | 10.2; *Ann. 5 | | | | | | | |
| | Chlorinated alkanes | | | | | 10.1; Ann. 4 | | | | | | | |
| | PBBs and PBDEs | 9.2; *Ann. 4 | | | | | | | | | | | |
| | PCDDs and PCDFs | | | | | | | | | | | *19.2 | |
| | PCDEs | | | 8.1; *Ann. 6 | | | | | | | | | |
| | TBT | 9.2; *Ann. 3 | | | | | | | | | | | |
| | Octachlorostyrene | | | | | | | | | 20.1 | | | |
| | Toxaphene | | | 8.1; *Ann. 5 | | | 12.3 | | | | | | |
| | Atrazine | | | | | | 12.1 | | | | | | |
| | Irgarol 1051 | | | 8.1; *Ann. 4 | | | | | | | | | |
| Classification/ assessment tools | Human health | | | | | | | | | | 19 | | |
| | Hazardous substances | | 11.1 | | | 12.3 | | | *15 | | | | |
| | Background concentrations | | | 15.1 | | 12.1 | | | | | | | |
| | Ecological Quality Objectives—North Sea | 17.2 | | | | | | | | | | | |
| | Ecotoxicolgical reference values | | | | | 12.2 | | | | | | | |
| | Environmental indicators | 17.1 | | | | | | | | | | | |
| Sand/gravel extraction | Code of Practice | | | | | | | | | | 16 | | |
| | Effects | 8.1 | 6.1 | *6.3 | | | | | | 15 | 16 | 15 | 15 |
| | Environmental impact assessment | 8.1 | | | | *15 | *15 | 13 | | | | | |
| Modelling | Radioactive contaminants/Baltic Sea | | | | | | | | *17.1 | | | | |
| | Use in monitoring and assessment | | | | | | 16 | | 17.2 | | | | |

| Topic | Sub-topic | 1999 | 1998 | 1997 | 1996 | 1995 | 1994 | 1993 | 1992 | 1991 | 1990 | 1989 | 1988 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Data banks and management | Nutrients | 19.2 | 15.2 | 17.2 | 16.1.2 | | | | | | | | |
| | Contaminants | 19.1 | 15.1 | 17.1 | 16.1.1; 16.3 | 17 | 2.2 | 2.2 | | | | | |
| | NSTF | | | | | | | | 20 | *21 | 22 | | |
| | ICES format | 19.1 | | | 16.6 | | | | | | | | |
| | ICES databases | 19.1 | | | | | | 14 | | | | | |
| | Biological database | 19.3 | 15.1.3 | 17.3; 17.4 | | | 11.2; Ann. 4 | | | | | | |
| | AMAP | 19.1.3 | | 17.1.1 | 16.2 | | | | | | | | |
| Ecosystem effects of fishing | General | | | *12 | 12 | | 18 | | *19 | 19 | | | |
| | Effects of disturbance on benthos | | 10.4 | 9.3 | 11.2 | 9; Ann. 3 | 11.1 | | 8.3 | 8.2 | | | |
| | Seabird/fish interactions | 4 | | 10 | | | 19 | | | | | | |
| | Changes in abundance of non-target fish species | | *13.3 | | | | | | | | | | |
| | Behaviour of community metrics | | 13.4.1 | | | | | | | | | | |
| | Effects on level of predation on benthos by fish | | 13.4.2 | | | | | | | | | | |
| | Impact on size/age and spatial distributions of target fish | | 13.1 | | | | | | | | | | |
| | Discards | | 13.2 | | | | | | | | | | |
| Inputs of contaminants and nutrients | Riverine inputs<br>• gross | | 4.7.2; 4.7.3 | *4.7.1 | | | | | | | | | |
| | • net | | | | | | | | | | | 16 | 16.2 |
| | Trend detection methods | 6; *Ann. 1 | | | | | | | | | | | |
| | Atmospheric inputs | | | 4.7.1 | | | | | | | | | 16.3 |
| ICES Environmental Report | Oceanographic conditions | 8.4.1 | 6.2.1 | | | | | | | | | | |
| | Zooplankton | | 6.2.2 | | | | | | | | | | |
| | Harmful algal blooms | 8.4.2; Ann. 2 | 6.2.3; Ann. 3 | | | | | | | | | | |

| Topic | Sub-topic | 1999 | 1998 | 1997 | 1996 | 1995 | 1994 | 1993 | 1992 | 1991 | 1990 | 1989 | 1988 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Special topics | Context of ACMP advice | | | | | | | | | | Ann. 7 | *21 | |
| | Patchiness in Baltic Sea | | | | | | | | | | | 17.1 | 19.1 |
| | Nutrient trends/eutrophication in OSPAR area | | | Ann. 9 | | | | | | | 13 | 12 | 11.1 |
| | Nutrients and eutrophication | | 10.1 | 9.1 | 9.1 | 5.8 | | 6.3 | 10 | *11.3 | | | 10.4 |
| | Sediments—Baltic | 5.2 | 4.3 | 6.1 | 7.1 | 7.1 | 7.1 | | | 14.1 | 15.1 | 14.2 | 19.3 |
| | Sediments<br>• bioavailability | | | | 9.3 | 4.2; Ann. 2 | 5.4 | 6.4; Ann. 2 | | 7.4 | | 7.3 | |
| | • release of contaminants | | | | | | | | | | | 14.3 | |
| | Bioaccumulation of contaminants | | | 8.2; *Ann. 7 | | | | | | | | | |
| | Acid rain studies/effects | | | | | | | | | | | 20 | 17 |
| | Coastal zone fluxes | | | | | | | 8.2 | | | | | |
| | Influence of biological factors on contaminant concentrations | | 7.2; Ann. 5 | | | | | | | | | | |
| | Discharge of produced water by offshore platforms | | 7.6; Ann. 7 | | | | | | | | | | |
| | North Sea Benthos Survey | | | | | 9 | | | *Ann. 5 | | | | |
| | GLOBEC | 18.1 | 16.2 | | | | | | | | | | |
| | GOOS | 18.2 | 16.1 | 16 | | | | | | | | | |
| | Marine habitat classification/ mapping | 16 | | | | | | | | | | | |

## ANNEX 10

## TITLES OF RECENTLY PUBLISHED ICES COOPERATIVE RESEARCH REPORTS

| No. | Title |
|---|---|
| 212 | Report of the ICES Advisory Committee on the Marine Environment, 1995 |
| 213 | Report on the Results of the Fifth Intercomparison Exercise for Nutrients in Sea Water |
| 214 | Report of the ICES Advisory Committee on Fishery Management, 1995 (Part 1 and Part 2) |
| 215 | Manual of Methods of Measuring the Selectivity of Towed Fishing Gears |
| 216 | Seabird/Fish Interactions, with Particular Reference to Seabirds in the North Sea |
| 217 | Report of the ICES Advisory Committee on the Marine Environment, 1996 |
| 218 | Atlas of North Sea Benthic Infauna |
| 219 | Database Report of the Stomach Sampling Project, 1991 |
| 220 | Guide to Identification of North Sea Fish Using Premaxillae and Vertebrae |
| 221 | Report of the ICES Advisory Committee on Fishery Management, 1996 (Part 1 and Part 2) |
| 222 | Report of the ICES Advisory Committee on the Marine Environment, 1997 |
| 223 | Report of the ICES Advisory Committee on Fishery Management, 1997 (Part 1 and Part 2) |
| 224 | Ballast Water: Ecological and Fisheries Implications |
| 225 | North Atlantic-Norwegian Sea Exchanges: The ICES NANSEN Project |
| 226 | Report on the Results of the ICES/IOC/OSPARCOM Intercomparison Programme on the Determination of Chlorobiphenyl Congeners in Marine Media—Step 3a, 3b, 4 and Assessment |
| 227 | Tenth ICES Dialogue Meeting |
| 228 | Report of the 11th ICES Dialogue Meeting on the Relationship between Scientific Advice and Fisheries Management |
| 229 | Report of the ICES Advisory Committee on Fishery Management, 1998 (Part 1 and Part 2) |
| 230 | Working Group on Methods of Fish Stock Assessment—Reports of Meetings in 1993 and 1995 |
| 231 | Status of Introductions of Non-Indigenous Marine Species to North Atlantic Waters 1981–1991 |
| 232 | Diets of Seabirds and Consequences of Changes in Food Supply |
| 233 | Report of the ICES Advisory Committee on the Marine Environment, 1998 |
| 234 | Report of the Workshop on Ocean Climate of the NW Atlantic during the 1960s and 1970s and Consequences for Gadoid Populations |
| 235 | Methodology for Target Strength Measurements (with special reference to *in situ* techniques for fish and micronekton) |
| 236 | Report of the ICES Advisory Committee on Fishery Management, 1999 (Part 1 and Part 2) |
| 237 | Seventh Intercomparison Exercise on Trace Metals in Sea Water |
| 238 | Report on Echo Trace Classification |

# ACRONYMS

| | |
|---|---|
| ACFM | Advisory Committee on Fishery Management |
| ACG | Assessment Coordination Group (OSPAR) |
| AChE | acetylcholinesterase |
| ACME | Advisory Committee on the Marine Environment |
| ACMP | Advisory Committee on Marine Pollution |
| ADI | acceptable daily intake |
| AHF | altered hepato fico |
| AHH | aryl hydrocarbon hydroxylase |
| ALA-D | δ-aminolevulinic acid dehydratase |
| AMAP | Arctic Monitoring and Assessment Programme |
| ANOVA | analysis of variance |
| ASC | Annual Science Conference (ICES) |
| ASE | accelerated solvent extraction |
| ASG | Assessment Steering Group (AMAP) |
| ASMO | Environmental Assessment and Monitoring Committee (OSPAR) |
| ASP | amnesic shellfish poisoning |
| ATHN | 7-acetyl-1,1,3,4,4,6-hexamethyltetrahydro-naphthalene |
| BCF | bioconcentration factor |
| BCPS | bis-*p*-chlorophenyl sulfone |
| BCR | European Commission Bureau of Community References |
| BEQUALM | Biological Effects Quality Assurance in Monitoring Programmes |
| BEWG | Benthos Ecology Working Group |
| BFG | Institute of Hydrology (Koblenz, Germany) |
| BMB | Baltic Marine Biologists |
| BMP | Baltic Monitoring Programme (HELCOM) |
| BRC | background reference concentrations |
| BSPAs | Baltic Sea Protected Areas |
| CAFF | Conservation of Arctic Flora and Fauna |
| CBs | chlorobiphenyls |
| CDEs | chlorodiphenylethers |
| CD-ROM | compact disc: read-only memory |
| CEFAS | Centre for Environment, Fisheries and Aquaculture Science (UK) |
| CFP | ciguatera fish poisoning |
| CHBs | chlorinated bornanes |
| CIEM | Conseil International pour l'Exploration de la Mer |
| CMDGC | comprehensive multidimensional gas chromatography |
| CMP | Cooperative ICES Monitoring Studies Programme |
| COMBINE | Cooperative Monitoring in the Baltic Marine Environment (HELCOM) |
| CPR | Continuous Plankton Recorder |
| CPUE | catch per unit effort |
| CRIMP | Centre for Research on Introduced Marine Pests (Australia) |
| CRMs | certified reference materials |
| CTD | conductivity–temperature–density |
| CUSUM | **Cumulative Sum** |
| CV | coefficient of variation |
| DBT | dibutyltin |
| DCM | dichloromethane |
| DDE | dichlorodiphenylethylene |
| DDT | dichlorodiphenyltrichloroethane |
| ΣDDT | total DDT |
| DG | Directorate General |
| DIFFCHEM | OSPAR Working Group on Diffuse Sources |
| DMA | dimethylarsenic |
| DNA | deoxyribonucleic acid |
| DO | dissolved oxygen |
| DOC | dissolved organic carbon |
| DPSIR | driving-forces-pressure-state-impact-responses |
| DR-CALUX | dioxin-responsive chemical-activated luciferase |
| DSP | diarrhetic shellfish poisoning |
| DST | diarrhetic shellfish toxin |
| DTA | direct toxicity assessment |

| | |
|---|---|
| DYNAMEC | Ad Hoc Working Group on the Development of a Dynamic Selection and Prioritisation Mechanism for Hazardous Substances (OSPAR) |
| EAC | ecotoxicological assessment criteria |
| EC | European Commission |
| EC MON | Environment Committee Working Group on Monitoring and Assessment (HELCOM) |
| ECD | electron capture detection |
| ECE LRTAP | Economic Commission for Europe Long-Range Transboundary Air Pollution Convention (UN) |
| EcoQO | ecological quality objective |
| EEA | European Environment Agency |
| EI | electron impact ionization |
| EIA | environmental impact assessment |
| ELISA | enzyme-linked immunosorbent assays |
| EMD | evaporative mass detector |
| ENDS | Environmental Data Services (UK) |
| ENSO | El Niño Southern Oscillation |
| EOC | elemental organic carbon |
| EOCl | extractable organic chlorine |
| EPA | Environmental Protection Agency (USA) |
| EQG | environmental quality guidelines |
| EQS | environmental quality standard |
| ER-CALUX | oestrogen-responsive chemical-activated luciferase |
| EROD | ethoxyresorufin-*O*-deethylase |
| ESB | Environmental Specimen Bank (Germany) |
| ESE | enhanced solvent extraction |
| ETC/MCE | European Topic Centre on Marine and Coastal Environment |
| EU | European Union |
| EUNIS | European Nature Information System |
| EUROSTAT | Statistical Office of the European Communities |
| EUT | Ad Hoc Working Group on Eutrophication (OSPAR) |
| FAO | Food and Agriculture Organization |
| FDE | Fish Disease Data Entry Program |
| FID | flame ionization detection |
| GABA | $\gamma$-aminobutyric acid |
| GC | gas chromatography |
| GC/ECD | gas chromatography/electron capture detection |
| GC/MS | gas chromatography/mass spectrometry |
| GESAMP | Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection |
| GEOHAB | Global Ecology and Oceanography of Harmful Algal Blooms (IOC-SCOR) |
| GPC | gel permeation chromatography |
| GLOBEC | Global Ocean Ecosystem Dynamics Programme |
| GMO | genetically modified organism |
| GnRN | gonadotropin-releasing hormone |
| GOOS | Global Ocean Observing System |
| GSI | gonadosomatic index |
| GST | glutathion-*S*-transferase(s) |
| HAB | harmful algal bloom |
| HAE | harmful algal event |
| HAEDAT | Harmful Algal Event Database |
| HARP | Harmonized Reporting Procedures (OSPAR) |
| HCB | hexachlorobenzene |
| HCH | hexachlorocyclohexane |
| H&E | haematoxylylin and eosin |
| HELCOM | Helsinki Commission (Baltic Marine Environment Protection Commission) |
| HHCB | 1,3,4,6,7,8-hexahydro-4,6,6,7,8,8-hexamethyl-cyclopental[*g*]-2-benzopyrane |
| HPLC | high performance liquid chromatography |
| HRMS | high resolution mass spectrometry |
| HTCO | high temperature catalytic oxidation |
| IAEA | International Atomic Energy Agency |
| IASC | International Arctic Science Committee |
| IBTS | International Bottom Trawl Survey |
| ICES | International Council for the Exploration of the Sea |
| ICS | International Chamber of Shipping |
| IEH | Institute for Environment and Health (UK) |
| IFREMER | Institut Français de Recherche pour l'Exploitation de la Mer |
| IJC | International Joint Commission (Canada and the USA) |

| | |
|---|---|
| IMO | International Maritime Organization |
| IMPACT | Working Group on Impacts on the Marine Environment (OSPAR) |
| INPUT | Working Group on Inputs to the Marine Environment (OSPAR) |
| IOC | Intergovernmental Oceanographic Commission |
| IOS | Initial Observing System (GOOS) |
| IOW | Institute of Baltic Research (Warnemünde, Germany) |
| ISO | International Organization for Standardization |
| ITIS | Interagency Taxonomic Information System (USA) |
| IUCN | International Union for the Conservation of Nature and Natural Resources |
| IWC | International Whaling Commission |
| JAMP | OSPAR Joint Assessment and Monitoring Programme |
| JBSAEM | Joint Meeting of WGBEC and WGSAEM |
| JGOFS | Joint Global Ocean Flux Study (IGBP) |
| JMP | OSPAR Joint Monitoring Programme |
| JNCC | Joint Nature Conservation Committee (UK) |
| LD | lethal dose |
| LMR | Living Marine Resources Panel (GOOS) |
| LOI | loss-on-ignition |
| LPS | Laboratory Performance Scheme (QUASIMEME) |
| LRMs | laboratory reference materials |
| LRMS | low resolution mass spectrometry |
| MA | Marketing Authorisation |
| MAFF | Ministry of Agriculture, Fisheries and Food (UK) |
| MarLIN | Marine Life Information Network (UK) |
| MAST | Marine Science and Technology Programme (EC) |
| MBT | monobutyltin |
| MCWG | Marine Chemistry Working Group |
| MDGC | multidimensional gas chromatography |
| MDR | multidrug resistance |
| MEDPOL | Monitoring and Research Programme of the Mediterranean Action Plan |
| MFO | mixed-function oxidase |
| MMA | monomethylarsenic |
| MMHg | monomethylmercury |
| MMP | Monitoring Master Plan (NSTF) |
| MON | Ad Hoc Working Group on Monitoring (OSPAR) |
| MPA | Marine Protected Area |
| mRNA | messenger ribonucleic acid |
| MS | mass spectrometry |
| MSD | mass selective detector |
| MSY | maximum sustainable yield |
| MT | metallothionein |
| mtDNA | mitochrondrial DNA |
| MXR | multixenobiotic resistance |
| NAO | North Atlantic Oscillation |
| NCI | negative chemical ionization |
| NCM | Nordic Council of Ministers |
| NIHS | National Institute of Health Science (Japan) |
| NIST | U.S. National Institute of Standards and Testing |
| NIVA | Norsk instittutt for vannforskning [Norwegian Institute for Water Research] |
| NMBAQC | National Marine Biological Analytical Quality Control Scheme (UK) |
| NMR | nuclear magnetic resonance |
| NOAA | National Oceanic and Atmospheric Administration (USA) |
| NOAEC | no observable adverse effects concentration |
| NOAEL | no observable adverse effects level |
| NODC | National Oceanographic Data Center (USA) |
| NOEL | no observable effects level |
| NORWECOM | Norwegian Ecological Model |
| NOWESP | North-West European Shelf Programme (EU MAST Project) |
| NRC | National Research Council (Canada) |
| NSP | neurotoxic shellfish poisoning |
| NSTF | North Sea Task Force |
| OCs | organochlorines |
| OCPs | organochlorine pesticides |
| OECD | Organisation for Economic Cooperation and Development |
| OIE | Office International des Epizooites |

| | |
|---|---|
| OM | oxidizable matter |
| OPs | organophosphates |
| OSPAR | OSPAR Commission |
| PAHs | polycyclic aromatic hydrocarbons |
| PAMP | Post-Authorisation Monitoring Programme (Scotland) |
| PAR | photosynthetic available radiation |
| PBBs | polybrominated biphenyls |
| PBDEs | polybrominated diphenylethers |
| PBTs | persistent, bioaccumulative, toxic compounds |
| PCA | principal component analysis |
| PCBs | polychlorinated biphenyls |
| PCDDs | polychlorinated dibenzo-*p*-dioxins |
| PCDEs | polychlorinated diphenylethers |
| PCDFs | polychlorinated dibenzofurans |
| PCNA | proliferating cell nuclear antigen |
| PCNs | polychlorinated naphthalenes |
| PCP | pentachlorophenol |
| PCR | polymerized chain reaction |
| PCTs | polychlorinated terphenyls |
| PEEK | polyetheretherketone |
| PFC | plaque-forming cell |
| PICES | North Pacific Marine Science Organization |
| PICT | pollution-induced community tolerance |
| PNEC | predicted no-effect concentration |
| POC | particulate organic carbon |
| POPs | persistent organic pollutants |
| PROD | pentoxyresorufin-*O*-deethylase |
| PSP | paralytic shellfish poisoning |
| PSU | practical salinity unit |
| PTFE | polytetrafluorethene |
| QA | quality assurance |
| QC | quality control |
| QSARs | quantitative structure–activity relationships |
| QSR | quality status report |
| QUASIMEME | Quality Assurance of Information for Marine Environmental Monitoring in Europe |
| QUASH | Quality Assurance of Sampling and Sample Handling (EC) |
| RIA | radioimmunoassay |
| RIKZ | Rijksinstituut voor Kust en Zee [National Institute for Coastal and Marine Management] |
| RMs | reference materials |
| ROSCOP | Cruise Summary Report |
| RUBIN | Rutin för Biologiska Inventeringar |
| SCOR | Scientific Committee on Oceanic Research |
| SFE | supercritical fluid extraction |
| SFG | scope for growth |
| SG | Study Group |
| SGBWS | ICES/IOC/IMO Study Group on Ballast Water and Sediments |
| SGFDDS | Study Group on Statistical Analysis of Fish Disease Data in Marine Fish Stocks |
| SGMBIS | Study Group on Marine Biocontrol of Invasive Species |
| SGMHM | Study Group on Marine Habitat Mapping |
| SGQAB | ICES/HELCOM Steering Group on Quality Assurance of Biological Measurements in the Baltic Sea |
| SGQAC | ICES/HELCOM Steering Group on Quality Assurance of Chemical Measurements in the Baltic Sea |
| SGQAE | ICES/OSPAR Steering Group on Quality Assurance of Biological Measurements related to Eutrophication Effects |
| SGSEF | Study Group on Effects of Sandeel Fishing |
| SIM | selected ion monitoring |
| SIME | Working Group on Concentrations, Trends and Effects of Substances in the Marine Environment (OSPAR) |
| SMLIPA | ICES Special Meeting on the Use of Liver Pathology in Flatfish for Monitoring Biological Effects of Contaminants |
| SOAEFD | Scottish Office Agriculture, Environment and Fisheries Department |
| SOPs | Standard Operating Procedures |
| SPM | suspended particulate material |
| SRMs | standard reference materials |
| TALs | total annual loads |
| TBA | tetrabutylammonium |

| | |
|---|---|
| TBPS | tertiary butylbicyclophosphorothionate |
| TBT | tributyltin |
| TCDD | tetrachlorodibenzo-*p*-dioxin |
| TDC | Thematic Data Centre |
| TEF | toxic equivalency factor |
| TIE | toxicity identification evaluation |
| TIMES | *ICES Techniques in Marine Environmental Sciences* |
| TIP | training and intercalibration programme |
| TMA | trimethylarsenic |
| TOC | total organic carbon |
| TON | total oxidized nitrogen |
| TPT | triphenyltin |
| UK | United Kingdom |
| UN | United Nations |
| UNEP | United Nations Environment Programme |
| UNCED | World Commission on Environment and Development |
| UNESCO | United Nations Educational, Scientific, and Cultural Organization |
| U.S. | United States |
| USA | United States of America |
| USEPA | United States Environmental Protection Agency |
| UV | ultraviolet |
| VIC | Voluntary International Contaminant Monitoring in Temporal Trends (OSPAR) |
| VMD | Veterinary Medicines Directorate (UK) |
| VPC | Veterinary Products Committee (UK) |
| VTG | vitellogenin |
| WGAGFM | Working Group on Application of Genetics in Fisheries and Mariculture |
| WGBAST | Baltic Salmon and Trout Assessment Working Group |
| WGBEC | Working Group on Biological Effects of Contaminants |
| WGBFAS | Baltic Fisheries Assessment Working Group |
| WGCCC | Working Group on Cod and Climate Change |
| WGEAMS | Working Group on Environmental Assessment and Monitoring Strategies |
| WGECO | Working Group on Ecosystem Effects of Fishing Activities |
| WGEIM | Working Group on Environmental Interactions of Mariculture |
| WGEXT | Working Group on the Effects of Extraction of Marine Sediments on the Marine Ecosystem |
| WGHABD | ICES/IOC Working Group on Harmful Algal Bloom Dynamics |
| WGITMO | Working Group on Introductions and Transfers of Marine Organisms |
| WGMDM | Working Group on Marine Data Management |
| WGMMHA | Working Group on Marine Mammal Habitats |
| WGMMPD | Working Group on Marine Mammal Population Dynamics and Trophic Interactions |
| WGMS | Working Group on Marine Sediments in Relation to Pollution |
| WGNAS | Working Group on North Atlantic Salmon |
| WGOH | Working Group on Oceanic Hydrography |
| WGPDMO | Working Group on Pathology and Diseases of Marine Organisms |
| WGPE | Working Group on Phytoplankton Ecology |
| WGSAEM | Working Group on the Statistical Aspects of Environmental Monitoring |
| WGSE | Working Group on Seabird Ecology |
| WGSSO | Working Group on Shelf Seas Oceanography |
| WGZE | Working Group on Zooplankton Ecology |
| WKBSED | ICES/HELCOM Workshop on Baltic Sea Sediments: Conditions and Contaminants |
| WKPHYT | ICES/HELCOM Workshop/Training Course on Phytoplankton |
| WKSMTD | ICES/OSPAR Workshop on the Identification of Statistical Methods for Trend Detection |
| WOCE | World Ocean Circulation Experiment |
| WWW | world wide web |